Cost-effective restoration measures that promote wider ecosystem and societal benefits

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[3] UHULL

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Summary

Water policies are often evaluated primarily on the basis of their financial (budgetary) costs, as these can be assessed relatively easily. The calculation of all costs and benefits, including second-order indirect effects on sectors and non-priced environmental effects, often also referred to as the broader social costs and benefits, is a more difficult task. Social cost-benefit analysis is a widely applied method for evaluating public water policies, since government interventions are often related to the provision of public goods, having an impact on society as a whole. Such impacts should consequently be valued and evaluated from a societal perspective, not the perspective of the investor only such as a central or local government or a private company. Restored or 'natural' river corridors typically have the potential to provide a wide range of ecosystem services. It is the wider social value attached to these ecosystem services besides their ecological value that is often missing in information supply supporting river restoration policy and decision-making.

The report provides an overview of existing guidelines and manuals related to the assessment of costs and benefits of river restoration. Although there exist many cost-benefit analysis handbooks, there are not many related specifically to river restoration. This report aims to fill this gap, and focuses on the specific characteristics of the estimation of costs and benefits related to river restoration. The report discusses the classification and assessment of costs and benefits of river restoration, and develops a benefits transfer approach that can be used to assess benefits when it is not possible to carry out primary valuation research. Key methodological issues in a CBA of river restoration are identified, discussed and illustrated. The report provides a number of practical recommendations to practitioners.

Keywords: River restoration, Cost-benefit analysis, Non-market valuation, Benefits transfer.

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Table of Contents

1 Introduction

The EU Framework 7 funded project Restoring Rivers for Effective Catchment Management (REFORM) aims to develop guidance and tools to ensure river restoration measures are cost-effective and support future River Basin Management Plans (RMBPs) for the European Water Framework Directive (WFD). This includes the analysis of the costs and benefits associated with river restoration projects. In practice, water policies are often evaluated primarily on the basis of their financial (budgetary) costs, as these can be assessed relatively easily. The calculation of all costs and benefits, including (second-order) indirect effects on sectors and (non-priced) environmental effects, often also referred to as the broader social costs and benefits, is a more difficult task. Social cost-benefit analysis (CBA) is a widely applied method for evaluating public water policies, since government interventions are often related to the provision of public goods, having an impact on society as a whole. Such impacts should consequently be valued and evaluated from a societal perspective, not the perspective of the investor only such as a central or local government (e.g. municipality). Restored or 'natural' river corridors typically have the potential to provide a wide range of ecosystem services. It is the wider social value attached to these ecosystem services besides their ecological value that is often missing in information supply supporting river restoration policy and decision-making.

CBA is carried out in order to evaluate and compare the various advantages and disadvantages of (alternative) river restoration projects in a structured and systematic way. The benefits from a restoration project are contrasted with the associated costs within a common analytical framework with clearly defined spatial and temporal boundaries. To allow comparison of these costs and benefits related to a wide range of impacts, measured in widely differing units, money is used as the common denominator. The results of this analysis can be interpreted as a B-C ratio, that is, total benefits divided by total costs, where a ratio larger than one indicates that the policy measure is beneficial from a social point of view and hence yields a welfare improvement. A CBA compares the costs and benefits of different restoration options in monetary terms. Strictly speaking, only those costs and benefits are included in a CBA that can be quantified in monetary terms. This is where usually most problems start for river restoration project appraisal since many effects, in particular ecological benefits, are often not priced in monetary terms. For many goods and services provided by restored or natural water resources, there is no market on which they are traded, and therefore no market price is available, which reflects their economic or social value. Hence, it will hardly ever be possible to monetize all impacts all the time. Those impacts that cannot be monetized are therefore often left out of the analysis.

While a textbook CBA requires that all impacts be monetized, in practice different approaches exist on how non-monetized impacts are included in the CBA. Non-monetized impacts, if considered relevant, can for instance be included in a qualitative discussion accompanying the CBA results. Pearce (1998) argues that in early CBA's conducted in the UK, such impacts would have been either ignored entirely, left for a subsequent environmental impact analysis, or monetized only partly. Applying an approach of monetizing impacts where possible, and including them in another form where monetization is not possible marks a deviation from the textbook ideal, but does not discredit the method as such. Moreover, there are nowadays several economic valuation methods, which allow placing a monetary value on non-marketed goods and services.

Including these non-market values in a CBA means that a wide range of environmental goods and services provided by river restoration are explicitly recognized in the CBA.

In this report, some of the key issues related to the assessment of the costs and benefits of river restoration projects will be identified and exemplified. Their relevance is, where possible, illustrated based on practical case studies. The key issues are structured following the general procedure in a CBA, illustrated in the box below.

The following general steps are typically followed in a CBA:

Step 1: Define the problem and objective of the policy action (in casu river restoration)

Step 2: Define the baseline scenario, i.e. what would happen if no action is taken

Step 3: Define the policy scenario(s) and alternative option(s) to achieve the objective

Step 4: Quantify the investment and running costs of each option

Step 5: Identify and quantify the positive and negative effects of each alternative option

Step 6: Value effects in money terms, using market prices and economic valuation methods

Step 7: Calculate the present value of costs and benefits occurring at different points in time

Step 8: Calculate the Net Present Value (NPV) or Benefit-Cost (B-C) ratio of each alternative

Step 9: Perform a sensitivity analysis

Step 10: Select the most beneficial policy action

Source: Brouwer and Pearce (2005).

It is important to point out that carrying out a CBA is a multi-disciplinary process, involving expertise from different fields and the input from policy and decision-makers. While economists are involved in all steps, environmental expertise of many kinds is also needed, especially in steps 2, 5 and 6. In order to ensure that the policy options are technically feasible, input from engineers is required especially in step 3, and often also in step 4 to specify the exact nature of the policy action or measure and estimate the associated investment and running costs. Policy and decision-maker input is essential when defining the objective the policy measures are supposed to achieve, and when defining the baseline and policy scenarios, including current policy. A key role of the economist in the whole process is to frame the relevant issues and develop the CBA framework so that all socio-economic stakes and stakeholders are included and the multitude of environmental studies that need to be undertaken are working towards answering the following two questions:

- Is river restoration economically speaking worthwhile, that is, do the benefits outweigh the costs?

- And if there are alternative river restoration projects available from which to choose, which river restoration project yields the highest net benefit?

The remainder of this report is structured as follows. Chapter 2 provides a brief overview of existing guidelines and manuals related to the assessment of costs and benefits of river restoration. Although there exist many cost-benefit analysis handbooks, there are not many related specifically to river restoration. This report aims to fill this gap, and focuses on the specific characteristics of the estimation of costs and benefits related to river restoration. The available data and information related to costs of river restoration is presented in Chapter 3 and the data and information related to benefits in Chapter 4. The key methodological issues in a CBA of river restoration are identified, discussed and illustrated in Chapter 5. These key methodological issues were identified by the REFORM project group during several project meetings, and are partly based on the work carried out in REFORM work packages 1 and 4. Conclusions and recommendations to practitioners are provided in Chapter 6.

2 Existing manuals and handbooks on the economics of river restoration

2.1 Introduction

This chapter provides a review of existing manuals and guidelines on the economics of river restoration. The chapter aims to provide an inventory and summarize the state of the art of current practices in conducting CBA of river restoration projects and identify the niche of the particular contribution of this REFORM report. The principle here is that the deliverable aims to contribute to the existing literature, and does not duplicate it. Where possible, links are established with existing manuals and handbooks. The search for existing manuals and handbooks was carried out using the web of science and scopus, as well as google for both published and grey literature. Alternative combinations of keywords relevant to the topics were used in the search with different search engines.

Based on this search, it became clear quite early on already that there exists no specific manual or guidance dedicated to the economics of river restoration. There also exists no journal article addressing specific economic issues of river restoration in a general way such as a literature review. However, there are many specific case studies on estimating the benefits of different types of river restoration such as dam removal and river channel modification. Also, there are quite a few number of edited books and guidance documents on CBA to support environmental policy in general (e.g. Hanley and Spash, 1993; Pearce et al., 2006), and a limited number particularly for water management. Below are three main examples related to water resources management (in chronological order):

- Brouwer, R., Pearce, D., 2005. Cost-Benefit Analysis and Water Resources Management. Edward Elgar Publishing, UK.
- Brouwer, R., Barton, D., Bateman, I., Brander, L., Georgiou, S., Martin-Ortega, J., Navrud, S., Pulido-Velazquez, M., Schaafsma, M., Wagtendonk, A., 2009. Economic Valuation of Environmental and Resource Costs and Benefits in the Water Framework Directive: Technical Guidelines for Practitioners. Project report.
- Shamier, N., Johnstone, C., Whiles, D., Cochrane, D., Moore, K., Lenane, R., Ryder S., Betts, V., Horton, B., Donovan, C., Harding, E., Bennett, R., Moseley, R., 2013. Water Appraisal Guidance: Assessing Costs and Benefits for River Basin Management Planning.

While all three references do not specifically address the costs and benefits of river restoration, they all touch upon a key component in the economic analysis of river restoration, namely the economic valuation of non-market benefits. Brouwer and Pearce (2005) is an edited book, which provides a solid foundation for the theory and methods of CBA of water resources management, along with case studies illustrating the practical aspects including for river restoration. The second reference by Brouwer et al. (2009) provides guidance on key issues in the economic valuation of water resources related to the implementation of the European Water Framework Directive (WFD) as part of the EU funded project AQUAMONEY. Here examples are included for water quality, ecological restoration of rivers and water scarcity. The third reference by Shamier et al. (2013) is particularly focused on the economic appraisal of policy interventions, projects, or programs for the purpose of supporting river basin planning management, a setting close to the economics of river restoration. The appraisal guidance is posted on the webpage dedicated to economics of the European Center for River Restoration (ECRR) (http://www.restorerivers.eu/RiverRestoration /Economics/tabid/2613/Default.aspx), although the report was developed for the UK Environment Agency. The review will focus more specifically on the second and third reference documents, attempting to shed more light on the practical aspects that this report could improve upon for conducting CBA of river restoration.

2.2 Economic Valuation of Environmental and Resource Costs and Benefits in the Water Framework Directive

This report, written as part of a specific targeted action under the 6th European Framework Programme addresses particularly economic valuation of environmental and resource benefits and costs for implementation of the WFD (i.e., related to reaching Good Ecological Status). It is targeted at expert practitioners or economic specialists with fundamental economic expertise and skills. It provides technical guidelines on key issues in economic valuation of non-market benefits and costs related to implementation of the WFD. The contents covered in the guidance include:

- Introduction
- Water valuation framework
- Aquatic ecosystem functions and total economic value
- Economic valuation methods
- Meta-analysis of non-market values for water services
- Water resource costs
- Water quality valuation: from WFD objectives to ecosystem goods and services
- Scale: from water body to river basin district
- Accounting for substitution effects
- Sensitivity to scope and procedural variance
- Payment certainty calibration
- Transfer errors
- Value aggregation
- Best practice recommendation

Figure 2.1 presents the organizational structure of the report, particularly on economic valuation and key issues, after introducing the theoretical foundation of water valuation, the ecosystem services approach to valuation, and valuation methods.

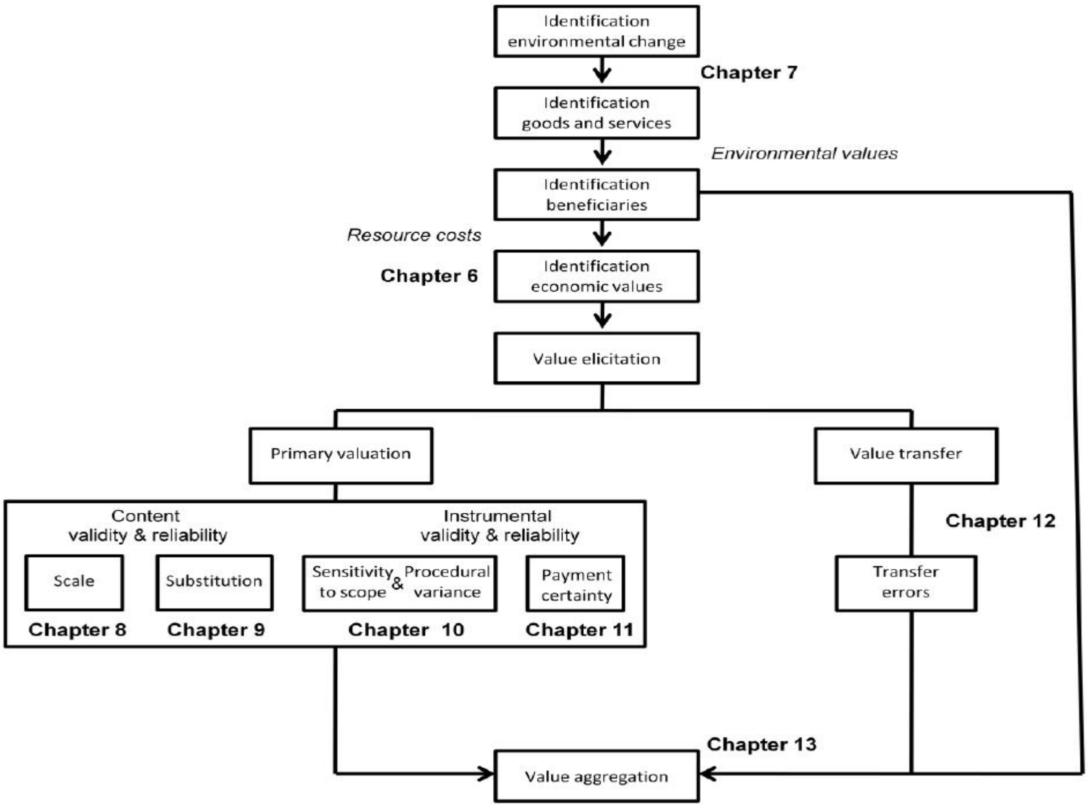


Figure 2.1. Organizational structure of economic valuation of environmental and resource costs and benefits for WFD (Source: Brouwer et al. 2009).

2.3 Water appraisal guidance

The water appraisal guidance provides a practical guideline on conducting CBA, non-market benefit assessment in particular, of water projects for river basin planning management. It adopts the ecosystem services approach to identify and quantify the benefits of water projects. The contents covered in the guidance include:

- Introduction
- The appraisal context
- Assessing the benefits - qualitative & quantitative assessment
- Assessing the benefits - stage 1 monetary valuation
- Assessing the benefits - stage 2 monetary valuation
- Comparing costs and benefits
- Sensitivity testing
- Final appraisal report

The development of the guidance strives to strike an appropriate balance and tradeoff between credibility (in terms of benefit value estimates) and usefulness (with respect to practical application without economic expertise), which is based on understanding of the technical challenges of non-market valuation and the needs of practitioners. Because of this special consideration and corresponding design with many built-in functions and data, the guidance is particularly suitable for policy analysts with a reduced requirement for economic expertise and skills. This implementation/application/operation-focused development approach is reflected in the procedure for benefit valuation and the forms and tables based assessment at different spatial scales with varying levels of detail and requirements of time and resources.

The guidance adopts a 5-steps procedure for valuing benefits (see Figure 2.2). The procedure starts with a qualitative description of the benefit brought by the considered project or program, followed by a quantitative description to the extent possible. The qualitative and quantitative description is largely a biophysical assessment of changes to ecosystems and benefits to stakeholders caused by the project intervention. Based on the description of benefits, stage 1 valuation assesses the monetary value of those benefits at the national level by linking them to the existing value estimates from the UK National Survey, which has developed a database of economic estimates for recreational, aesthetic, and non-use values. If the estimated benefit is greater than the (pre-determined) cost of the considered project, then the benefit valuation stops here. Note that the costs of projects or the estimation thereof are not covered in the guidance.

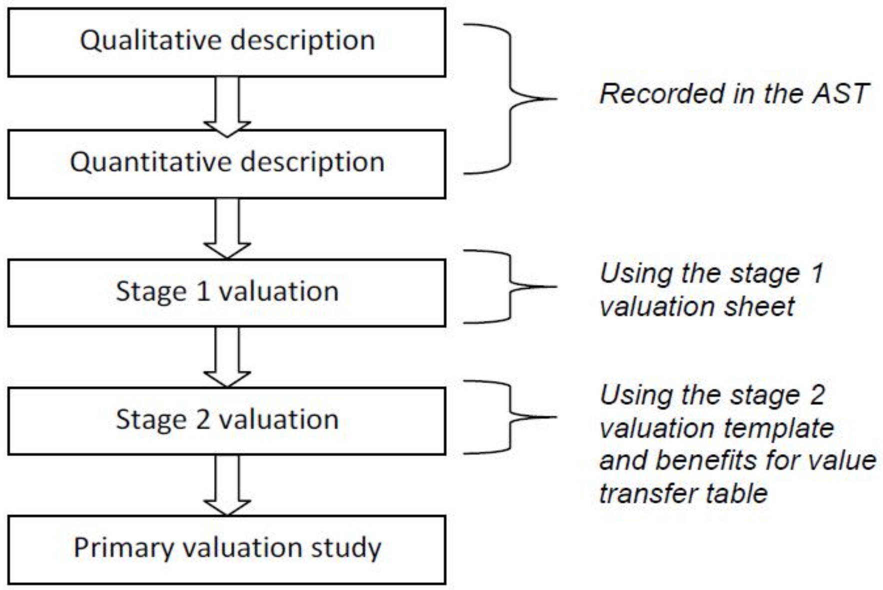


Figure 2.2. Procedure for economic valuation of benefits of water projects (Source: Shamier 2013).

In cases where the estimated benefit from stage 1 valuation is smaller than the (pre-determined) cost, further investigation of the economic benefit may be needed in stage 2. Stage 2 valuation uses benefits transfer to develop locally specific value estimates with more details based on a broader source of value estimates for a wider range of benefits from previous studies. This is a more structured economic valuation of non-market benefit without requiring a high level of economic expertise, and thus is still doable by non-economist specialists. In this stage, if the estimated benefit is greater than the cost, then the benefit valuation stops here with a corresponding conclusion in the cost-benefit appraisal. Otherwise, a primary valuation study would be needed for the appraisal, which requires economic expertise.

One significant feature of the guidance is its forms and tables based assessment at different spatial scales with varying details and requirements of time and resources. Specifically, the guidance developed a spreadsheet template called Appraisal Summary Table (AST) to assist in describing qualitatively and quantitatively the (biophysical) impacts of water projects (see Figure 2.3).

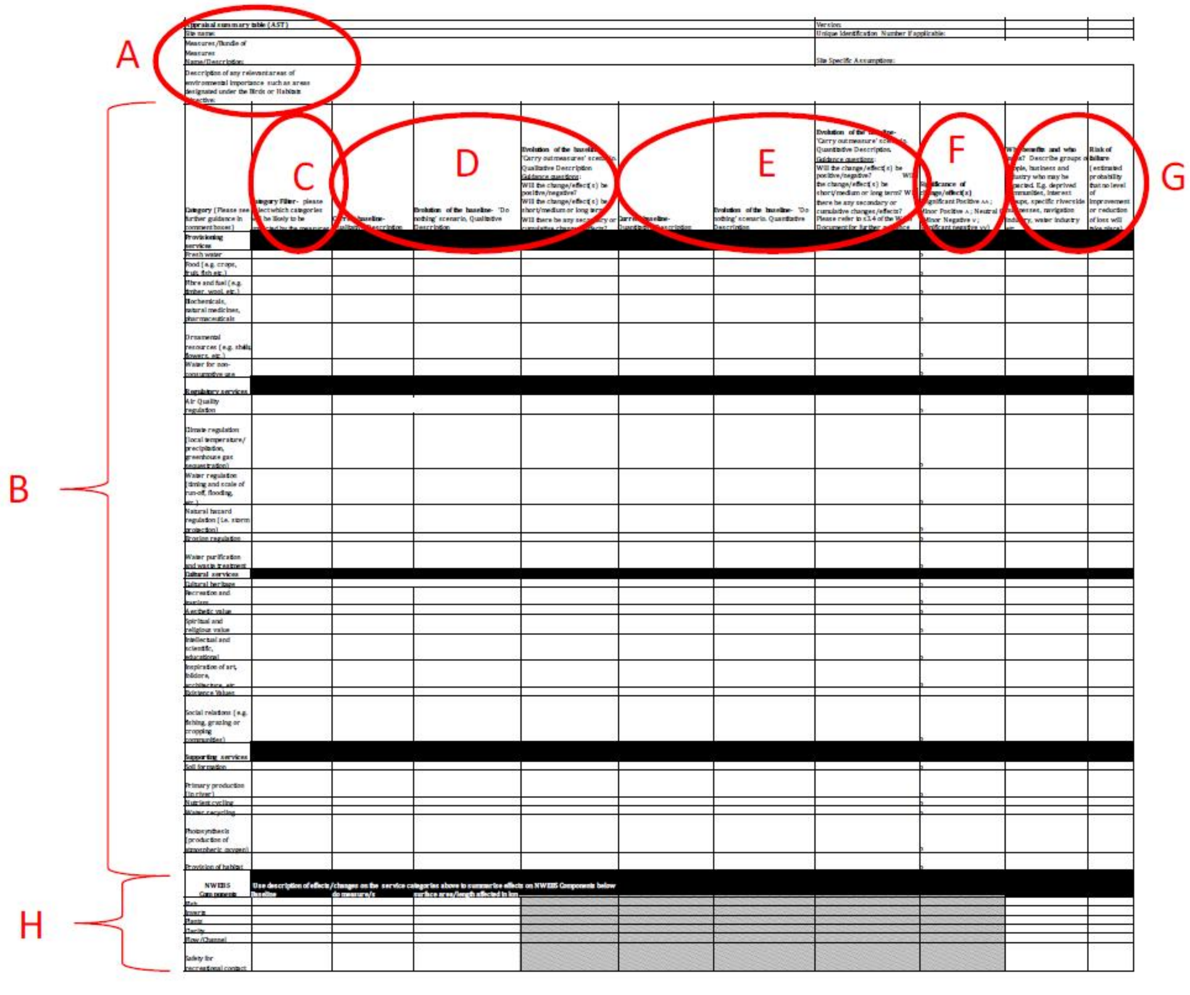


Figure 2.3. Illustration of appraisal summary table for impact assessment (Source: Shamier 2013).

The description of each item in the AST is listed below:

A. Project background information

B. Ecosystem services categories

C. Category filter

D. Qualitative description (current baseline, evolution of baseline - do nothing versus project intervention)

E. Quantitative description

F. Significance of change

G. Beneficiaries & effectiveness

H. National water environment benefits survey components

As demonstrated in Figure 2.3, the qualitative (and quantitative) impact assessment of a project intervention described by A is conducted within the framework of ecosystem services laid out by B. Items B-G in the table identify those impacts that are significant in terms of ecosystem services change, beneficiaries, and the effectiveness of the intervention. Item H lists components in the national water environment benefits survey that was conducted previously in the context of the WFD and for which value estimates for a limited number of benefits are available.

Figure 2.4 describes those benefits with available estimated value information as compared to different sets of benefits. It shows that the assessed benefit in this stage only represents a small portion of the full set of possible benefits. Consequently, stage 2 monetary valuation is needed to consider more benefits even if the estimated benefit value in stage 1 is smaller than the cost.

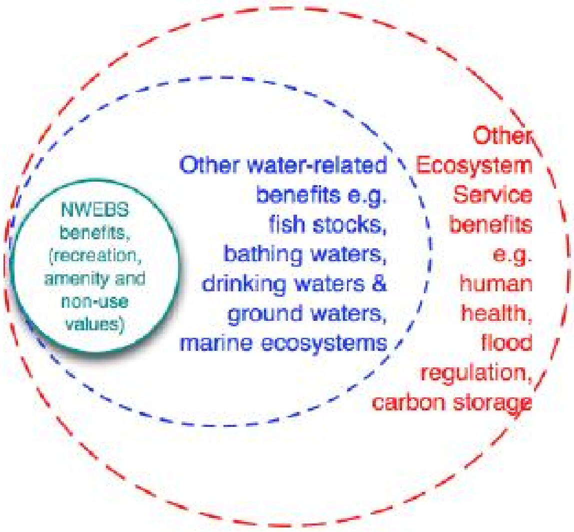


Figure 2.4. Illustration of benefits considered in the National Water Environment Benefits (Source: Shamier 2013)

	FISH	INVERTS	PLANTS	CLARITY	FLOW /CHANNEL	SAFETY
Pressure List						
2,4-dichlorophenoxyacetic acid	1	1	1	0	0	0
24d	1	1	1	0	0	0
Acid Neutralising Capacity	0	1	1	0	0	0
Acidification	1	1	0.5	0	0	0
Aldrin, Dieldrin, Endrin & Isodrin	1	1	1	0	0	0
Alien species-Chinese mitten crab	0	0	0	0	0	0
Alien species-fish stocking	1	0	0	0	0	0
Alien species-Floating pennywort	1	1	1	0	1	0
Alien species-North American Signal crayfish	0	1	0	0	0	0
Alien species-other crayfish	0	1	0	0	0	0
Alien species-other invertebrates	0	1	0	0	0	0
Alien species-other plants	0	0	1	0	0	0

Figure 2.5. Illustration of linking ecosystem service impact to benefit components (Source: Shamier 2013).

6. Should the measure/bundle of measures improve the condition of the waterbody or are they for no deterioration?	7. How many indicators of water quality are improved/have not deteriorated due to the measure/bundle of measures? (see Guidance sheet for list of indicators)	8a. The number of km improved or not deteriorated with respect to fish						8b. Willingness to pay value to be used for fish (default should be 'central') **only one box must be crossed**			9a. The number of km improved or not deteriorated with respect to **other animals such as invertibrates**					
	1-6	*Bad-Poor*	*Poor-Mod*	*Bad-Mod*	*Mod-Good*	*Poor-Good*	*Bad-Good*	*low*	*central*	*high*	*Bad-Poor*	*Poor-Mod*	*Bad-Mod*	*Mod-Good*	*Poor-Good*	*Bad-Good*
improvement	6				10				X					10		

9b. Willingness to pay value to be used **for other animals such as invertibrates** (default should be 'central') **only one box must be crossed**			10a. The number of km improved or not deteriorated with respect to **plant communities**						10b. Willingness to pay value to be used **for plant communities** (default should be 'central') **only one box must be crossed**			11a. The number of km improved or not deteriorated with respect to the **clarity of water**						11b. Willingness to pay value to be used for the **clarity of water** (default should be 'central') **only one box must be crossed**		
low	*central*	*high*	*Bad-Poor*	*Poor-Mod*	*Bad-Mod*	*Mod-Good*	*Poor-Good*	*Bad-Good*	*low*	*central*	*high*	*Bad-Poor*	*Poor-Mod*	*Bad-Mod*	*Mod-Good*	*Poor-Good*	*Bad-Good*	*low*	*central*	*high*
	X					10				X					10				X	

Figure 2.6 Illustration of valuation spreadsheet used in stage 1 (Source: Shamier 2013)

As mentioned above, stage 2 represents a more complete valuation with local details using benefits transfer guided by sensitivity analysis. Figure 2.7 illustrates what and how value estimates are used with benefits transfer for the specific benefit components identified from AST. The range of value estimates provides a foundation for conducting sensitivity analysis with alternative value estimates from different sources.

The physical characteristics of the goods: e.g. the impact, pollutant, habitat, species, resources, etc	Unit Value (£s)	Parameter (e.g. per household per annum	Value year	Source
Inland Marsh	~£1300, with range £200-£4300	per ha per annum	2008	EVEE guidance table 2.2.
Salt Marsh	~£1400 with range £200-£4500	per ha per annum	2008	EVEE guidance table 2.2.
Intertidal Mudflat	~£1300, with range £200-£4300	per ha per annum	2008	EVEE guidance table 2.2.
Peat Bog	~ £300 with range £0-£1000	per ha per annum	2008	EVEE guidance table 2.2.
Commercial fisheries; trout fingerlings	£170	per tonne restocking rate.	2002	Nix (2002) as summarised in table 4.3 in BAG part 2
Commercial fisheries; trout fry	£210	per tonne restocking rate.	2001	ABC (2001) as summarised in table 4.3 in BAG part 2
Commercial fisheries; Grisle (young) salmon	£2,200	per tonne restocking rate.	2001	ABC (2001) as summarised in table 4.3 in BAG part 2
Commercial fisheries; pre-salmon	£2,300	per tonne restocking rate.	2001	ABC (2001) as summarised in table 4.3 in BAG part 2
Commercial fisheries; salmon	£2,400	per tonne restocking rate.	2001	ABC (2001) as summarised in table 4.3 in BAG part 2
Commercial fisheries; Carp (fry, 2 to 3 inches long, 700 to 800 per acre)	25p	per fish restocking rate	2001	ABC (2001) as summarised in table 4.3 in BAG part 2

Figure 2.7. Illustration of value estimates for benefit transfer in stage 2 (Source: Shamier 2013).

2.4 Comparison of the existing guidelines

Both guidances share some similarities with regards to the following aspects:

- Economic valuation adopts the ecosystem service framework;
- Economic valuation is characterized by a structure of impact/change assessment followed by benefit valuation;
- River restoration is however not the major purpose for valuation;
- Financial and economic costs of restoration measures are not incorporated.

Both documents also differ from each other in the following aspects:

- Development context;

The appraisal guidance addresses benefit valuation (in relation to pre-determined costs) of water intervention for the purpose of supporting river basin planning management.

The technical guidance addresses economic valuation of environmental and resource costs and benefits in relation to implementation of the WFD.

- Targeted user group

The appraisal guidance is targeted at practitioners such as policy analysts or evaluation specialists without economic expertise.

The technical guidance is targeted at expert practitioners and economic specialists with some economic foundation.

- Purpose and guiding principles

The appraisal guidance is focused more on the usability and accessibility of valuation in practice, with explicit consideration of the valuation contexts and application aspects and the tradeoff between usability and credibility.

The technical guidelines are more focused on the validity and reliability of the value estimates, touching on key issues in economic valuation related to WFD, the only unique guidance particularly on water quality valuation, substitution effects, and aggregation over river basins.

Based on the above comparison, both documents can complement each other with the following implication for this particular report:

- The framework of water appraisal guidance for CBA can be adopted for economic valuation of the costs and benefits of river restoration if being applicable and user friendly are the major purposes with a targeted user group of non-economist professionals and practitioners.
- The database of benefits and valuation information built in the water appraisal guidance derived from the UK National Survey can be extended to cover a broader set of benefits spatially explicit at the global level, particularly for river restoration (see Chapter 4).
- The key issues addressed by the technical guidance can be explicitly accounted for or embedded in developing a form and table based appraisal of river restoration (see Chapter 5).

3 Reporting and predicting costs of river restoration projects

3.1 Economic analysis in the context of river restoration projects

Knowing the economic costs of hydromorphological restoration measures is undeniably important for planning cost-effective conservation schemes that achieve the greatest positive ecological impacts with a given budget. From an economic perspective, an evaluation of the varying economic costs of restoration is equally as important as identifying where the restoration measures will be most effective. Although an economic analysis is only one way to go about prioritising restoration projects, it can yield the most efficient restoration outcomes when watershed assessments provide necessary information on ecological pressures and the costs and benefits of proposed measures.

The Water Framework Directive (WFD) foresees economic analysis not only as underpinning for the selection of measures, but also for the justification of exemptions, so-called derogations (i.e. postponing the adoption of measures). Article 4 of the WFD cites "disproportionate costs" as a justification for not reaching targets within the foreseen timeframe. Although the perception of disproportionate costs can vary across Member States, an economic analysis should be undertaken to determine whether costs are indeed disproportionate.

A cost-effectiveness analysis would be sensible for selecting which measures should be implemented at the basin scale to achieve the good ecological status (GES) or good ecological potential (GEP) targets set forth in the WFD. Additionally, when considering the implications for an individual firm, a water body, or a river basin, this analysis can underpin the justification for time-scale derogations. Specifically, certain measures could be postponed in order to allow for new abatement techniques to be developed that lowered the total costs of abatement – or restoration, in this case. If, however, the costs are disproportionate for reasons other than financial viability, i.e., if the costs of the proposed measure are perceived to outweigh the benefits of reaching GES, then a standard-setting derogation could come into play. In such a case, a cost-benefit analysis at the margin would be necessary to identify a new optimal level of abatement (i.e. the point after which marginal costs begin to surpass marginal benefits). Specifically, if the GES standard is too restrictive, the social benefits of some of the marginal abatement options (e.g., hydromorphological restoration measures) implemented in order to reach it may actually be outweighed by their private or societal costs.

In summary, these different types of economic analysis, applied at various scales, can help to inform river management bodies in several ways. First of all, the programme of measures (PoM) is designed to list the measures being taken to reach an environmental target, namely the good ecological status of the relevant water bodies. As such, a cost-effectiveness analysis is suitable for managing resources efficiently. Additionally, the justification of disproportionate costs can rely on cost-effectiveness analysis to show that the achievement of the goal is not currently financially viable or cost-benefit to show that the marginal benefits of abatement are outweighed by marginal abatement costs at some point short of the good ecological standard.

3.2 The socio-economic costs associated to river restoration

Almost 50% of water bodies across 23 of the EU Member States are considered heavily impacted by hydromorphological alterations, with approximately 88% of these exhibiting hydromorphological degradation as a result (ETC/ICM, 2012). The major subcategories of these alterations that are present in Europe include (1) changes to the hydrological regime, including water impoundment by dams and other changes due to weirs and locks and (2) other river management practices such as dredging, land drainage, and the construction of barriers that directly affect the hydromorphological status of the watercourse. There are approximately 7,000 large dams in Europe and thousands of other smaller impoundments. Some waterways are impacted by these alterations in the extreme; for example, 91% of the water bodies in the Elbe River Basin in Germany fail to achieve GES due to hydromorphological pressures (ETC/ICM, 2012).

The benefits of altering and managing rivers accrue to society at large through the economic goods and services that these support. Three major industries or economic sectors that benefit from these alterations are agriculture (and other land uses that contribute to land drainage or the reclamation of active floodplains), transport over inland waterways, and hydroelectricity production. Although a comprehensive and accurate picture of the benefits that these economic sectors accrue through the alteration and subsequent degradation of some European waterways does not exist, a selection of economic indicators can provide context to the discussion of river alteration. For example, although transport on inland waterways only accounted for a mere 6.5% of total freight transport in the EU in 2010 (Eurostat, 2013), it is a competitive mode of inland freight transport that will likely have to grow in the future if the EU is to experience carbon emissions reductions in transport. The sector currently enjoys modal shares of up to 30% for bulk commodities (CE Delft, 2011). Transport on inland waterways produced approximately 8 billion Euros of gross value-added in the EU in 2007 and employed over 35,000 people (Ecorys, 2012). Meanwhile, agriculture produced over 405 billion Euros in value in 2012 (Eurostat, 2013). Finally, hydroelectric power accounts for 16% of electricity production and 70% of all renewable energy production in Europe (ETC/ICM, 2012). As such, it plays a major role in powering the decarbonisation of Europe's electricity sector, although most capacity has already been exploited (Kumar et al., 2011).

Clearly, these industries and the European economy as a whole depend on some river alteration to perform their activities, and the gains for these sectors and their consumers are significant. However, the costs of river alteration and degradation must also be weighed against these benefits. The socio-economic costs of hydromorphological alteration are a result of changes in the quantity and quality of water provided by rivers as well as barriers for migrating species caused by changes in the river structure. These changes may affect ecosystems, human health, and economic activities along the river. By estimating changes in production, costs of replacement, hedonic prices and by applying contingent valuation or an ecosystem services approach, the scope of these costs can be determined ex post.

3.3 The role of economic assessments in water policy: CBA and CEA

The concern about the integrity, resilience, and sustainability of river ecosystems has turned river restoration into a multi-billion dollar, global industry (Palmer et al., 2005). In its most formal sense, the term restoration refers to returning an ecosystem to its original pre-disturbance state; but, in practice, river restoration is used to refer to habitat enhancement, rehabilitation, improvement, mitigation, creation, and other situations (Roni et al., 2005). Some common goals of river restoration are to (i) improve water quality, (ii) re-establish river type-specific habitats and ecosystem functioning, (iii) aid in species recoveries, and (iv) maintain the provision of ecosystem services. Because decisions about river rehabilitation are societal ones, restoration projects that consider human dimensions (e.g., society's need for ecosystem services, conflicting interests of multiple stakeholders, and interactions of environmental policy, economics, and science) are more likely to meet environmental management and policy goals.

Ecological boundaries such as river basins do not conform to political and cultural boundaries, so solving water resource issues requires international understanding and cooperation. While the WFD's river basin approach should allow for increased comprehensiveness in water resources management by expanding it to include other policy areas such as land use, flood risk mitigation, navigation, hydroelectric power production, and nature conservation, approaches for integrating these governance responsibilities within river basins and across borders are left to the Member States. Of concern for river restoration is the interplay between hydromorphological quality parameters and these other policy areas, including land use, navigation, and dam operation. The achievement of GES and GEP thus depends on the ability of river basin managers to balance the needs of the WFD with those of these other fields effectively (Moss, 2004).

Balancing such concerns in a transparent manner requires an economic analysis of the impacts of these measures. River basin managers and authorities responsible for the implementation of measures to achieve the WFD GES/GEP goals are challenged to prioritize measures to efficiently use limited budgets while obtaining the greatest ecological and economic returns from these investments. Achieving environmental policy and management objectives to rehabilitate the degraded physico-chemical, hydromorphological, and biological elements of rivers requires the implementation of effective restoration measures, and the need to identify and evaluate these measures is growing (Kail and Wolter, 2011). During the 1st Management Cycle of the WFD, the majority of reported RBMPs did not describe the financial commitment, the responsible parties for implementation, the planned timetable, or the expected status improvements to result from the PoMs (European Commission, 2012). This lack of information hinders the achievement of the WFD not only by making it more difficult to assess whether sufficient action is being taken, but also by not providing a basis to determine whether restoration resources are being used effectively. For the implementation of the WFD, a cost-effectiveness analysis of measures can help to ensure that the least-cost options for achieving GES/GEP are chosen for the PoM (Lago, 2008). Ideally, such optimization would occur in a river basin setting and not be limited to the scale of individual measures.

Tools are needed that will allow decision makers and stakeholders to assess restoration measures better ex ante. Only by assessing the full spectrum of costs and benefits can decision makers effectively allocate public and private funds and ensure the best ecological outcomes. A framework for this assessment will need to inform the creation of the second round of RBMPs. Although predicting ecological responses is of obvious importance, an economic consideration of costs and benefits is essential for rationally managing our rivers. Introducing economics as a tool for the planning, prioritization, and evaluation of restoration projects is still in its infancy (Robbins and Daniels, 2012; Naidoo et al., 2006). In a meta-analysis of 1,582 recent peer-reviewed papers dealing with ecological restoration, Aronson et al. (2010) found that restoration scientists and practitioners are failing to show the links between the socio-economics and ecology of restoration, underselling the evidence for restoration as a worthwhile environmental and societal investment. While broad overviews of restoration prioritization for river basin managers and practitioners are available in the published literature (e.g., Roni et al., 2002; Beechie et al., 2008; Roni et al., 2008), a rationalized economic analysis to guide decisions and investments in restoration measures and to elicit the greatest impact (i.e., socio-economic and environmental benefits of restoration measures) is needed.

The proper assessment of the costs and benefits related to the implementation of river restoration measures forms the basis for effective river restoration management. By reviewing river restoration projects across the EU, Deliverable 1.4 of REFORM[1] found that in many cases costs had not been assessed in a structured way, thus hampering effective decision-making based on economic assessments, particularly cost-benefit analysis and cost-effectiveness analysis. The following section will thus discuss the proper design and implementation of cost assessments in the context of river restoration.

A typology of costs related to river restoration

This section will introduce the concepts of water use, value and costs as used under the EU Water Framework Directive, thereby outlining the relevance of cost assessment in the overarching policy framework. Furthermore, the costs related to hydromorphological restoration will be specified. The section will conclude by recommending a typology of costs which can form the basis for a cost assessment, particularly in the context of a cost-effectiveness analysis of alternative restoration measures.

The WFD ‘total cost’ concept

The WFD specifies a series of reporting dates for key tasks and activities aimed at the development and implementation of river basin management plans (see timetable for implementation in table 3.1), this applies to many elements of the Directive, including its economic requirements. However, the Directive’s legal text is a prescriptive document and does not clearly specify how to implement or develop its requirements and key elements. In consequence, the European Commission established informal working groups at European level to develop guidance documents to aid different aspects of the implementation process of the Directive, with the main objective of harmonising the implementation process across Europe and encouraging application (e.g. guidance documents have been produced on the analysis of pressures and impacts for the environmental characterisation documents or in the establishment of water quality standards).

[1] D1.4 Inventory of the cost of river degradation and the socio-economic aspects and costs and benefits. Available at http://www.reformrivers.eu/deliverables/d1-4

Table 3.1: WFD detailed timetable for implementation

Year	Issue	Reference
2000	Directive entered into force	Art. 25
2003	Transposition in national legislation	Art. 23
	Identification of River Basin Districts and Authorities	Art. 3
2004	Characterisation of river basin: pressures, impacts and economic analysis	Art. 5
2006	Establishment of monitoring network	Art. 8
	Start public consultation (at the latest)	Art. 14
2008	Present draft river basin management plan	Art. 13
2009	Finalise river basin management plan including progamme of measures	Art. 13 & 11
2010	Introduce pricing policies	Art. 9
2012	Make operational programmes of measures	Art. 11
2015	Meet environmental objectives	Art. 4
2021	First management cycle ends	Art. 4 & 13
2027	Second management cycle ends, final deadline for meeting objectives	Art. 4 & 13

Source: European Commission (http://ec.europa.eu/environment/water/water-framework/timetable.html)

The working group dedicated to the attention of the Directive's economic issues was set up in December 2000 and named WATECO (for water economics). Prior to 2013, this group only produced one official guidance document (European Commission, 2002), which covered general aspects of the economic analysis for the development of river basin management plans that were presented in 2008, paying special attention to the economic characterisation of river basin districts.

The WATECO group recognized that the economic analysis is a process of "*providing valuable information to aid decision-making and should be an essential part of the overall approach for supporting decisions*" (European Commission, 2002). In theory, the objective of the analysis is to serve as an exercise in the elicitation of trade-offs and it is to be undertaken in co-ordination with other types of information and input, such as from the public participation processes (Kallis, 2005).

By the end of 2004, it was required that each Member State undertook an economic analysis of water use for each of their river basins (see timetable for implementation in table 1). This was produced together with a preliminary assessment of the balance of demand and supply of water services and the pressures and impacts on the water environment[2]. In other words, the economic analysis should provide information on what it costs, who pays, who gains and who suffers from the current situation and has to be integrated with other technical analyses such as the environmental analysis of pressures and impacts. This aims to ensure that a common description and characterisation of the river basin is obtained and used as the basis for the identification of the programme of measures and the development of the river basin management plans. The results of the economic analysis will be used to inform future WFD-related decisions.

[2] For further information; the results of the analysis for each member state can be found at: http://forum.europa.eu.int/Public/irc/env/wfd/library?l=/framework_directive&vm=detailed&sb=Title

The concepts of water use, value and costs

For the achievement of sustainable uses of water resources, the Directive goes beyond the concept of water demand management (an instrument traditionally applied in water resources management which aims to attain optimal uses of water to ensure the financial sustainability of the service) and promotes the introduction of water pricing policies, which also account for the recovery of environmental and resource costs of the different types of use. This could be seen as a way to attain a level of sustainability of water use, more in accordance with the environmental objectives of the Directive.

The theory behind demand management of water services is fairly simple. It aims to attain some sort of economic optimality in use by taking into account the value of water in relation to the financial costs of provision[3] (Winpenny, 1994). In the context of the WFD, the objective is not only to achieve sustainable management of water resources but also sustainable uses. In consequence, one of the first steps in the economic analysis of water use is the identification of the different types of uses of water; each different use would imply a different economic value and in many instances also may incur costs. Unlike other commodities, the special characteristics of water as a resource, imply that the same good in theory has different economic uses in practice (which in the case of water, can also differ in levels of quantity and quality).

The typology of the different uses of water is a contested issue in the water economics literature. As an example, table 2 introduces a selection of the various classifications used in water economics to describe the different types of values associated with the goods and services provided by water resources. Again, these differences are related with the versatility of the resource, which introduces different points of view.

[3] In theory, this will be achieved when the marginal unit of water for each user has the same value.

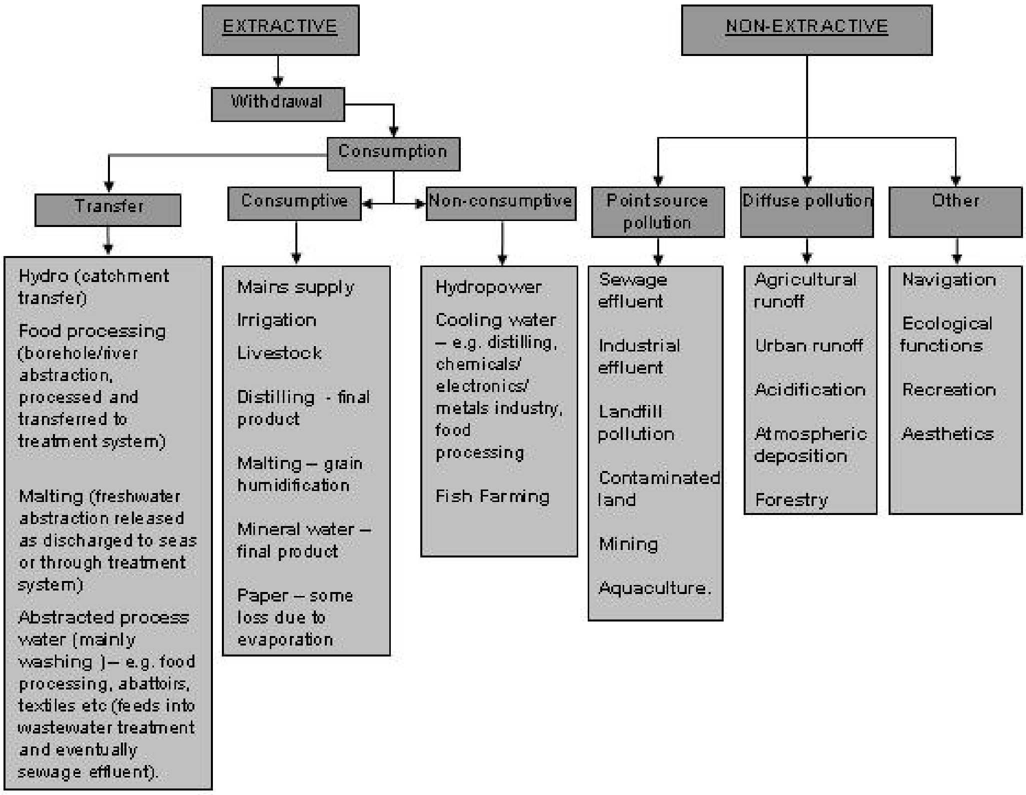
Table 3.2 Selected classifications of the value of water in economics (Modified from: Turner, Georgiou, Clark and Brouwer (2004)

Turner, Georgiou, Clark and Brouwer (2004)	Rogers, Bhatia and Huber (1997)	Young (1996)	De Groot (1992)
Describe the components of the value of water using conventional categories of Total Economic Value, which is the sum of: ▪ Direct use values: Arise from direct interaction with water resources. They can be consumptive, e.g. irrigation or non-consumptive, e.g. recreational swimming ▪ Indirect use values: Services provided by water resources but that do not entail direct interaction, e.g. flood protection by wetlands ▪ Non-use values: Existence, bequest and philanthropic value. ▪ Option value: Satisfaction of knowing that the resource is available to future generations ▪ Quasi-option value: Derived from the potential benefits of delaying action until further information is available, e.g. value placed on conservation of a wetland until further information is available on the value of the species that are found within it.	Value of water use comprises economic and intrinsic value: ▪ Value to other users: Value of water in industrial and agricultural use and WTP for its domestic use ▪ Net benefits of return flows: Recognises the vital role played by return flows in many hydrological systems e.g. recharge of aquifers ▪ Net benefits from indirect use: Benefits associated with improvements in income and in health that can accompany schemes that provide water for irrigation, domestic and livestock use. ▪ Adjustments for social objectives: e.g. poverty alleviation, employment generation or food security Intrinsic value of water: Includes the stewardship, bequest, and pure existence value	Water related economic values are divided into the following classes: ▪ Commodity benefits: These are derived from personal drinking, cooking and sanitation, and from productive economic activity, e.g. agriculture ▪ Aesthetic and recreational values ▪ Waste assimilation benefits: These result from the sink function of waterbodies that carries away residuals from processes of human production and consumption. ▪ Dis-benefits or damages: These are found in connection with evaluations of foodplain and water quality management. ▪ Non-use values from knowing that a good exists, even though no direct experience is had of the good. Other possible values, include: intrinsic, ecosystem preservation and socio-cultural.	Value is categorised in terms of the nature of the contribution made to human welfare, categories: • Ecological value: Includes conservation and existence values. Usually only described qualitatively as valuation is limited, though it may be described using quantitative indicators (e.g. number of species) • Social value: Includes health and option values. It may be quantified through use of minimum standards for resource availability (e.g. to ensure sustainable harvesting) • Economic values: Includes consumptive use, productive use and employment value. It can be described in monetary units (e.g. value of the resource harvested), quantities (e.g. volume of a resource harvested) or the number of the people employed in the activity

Water uses are often divided, in relation with their economic values (or benefits derived from its use) and the nature of the use. As the most typical example used in the literature, the total economic value approach divides water use into two main types: use and non-use values (Turner et al., 2004).

The Directive is concerned mainly with use values, which can be classified in: 1) direct use values, which are extractive and consumptive and have a direct impact on water quality and quantity. Some examples of direct uses are agricultural irrigation, water stored for hydroelectricity generation, drink production, etc... and 2) indirect use values, which are related with recreational and aesthetic uses, they are typically non extractive and non-consumptive. Figure 1 offers some examples of the different definitions of water use according to the WFD.

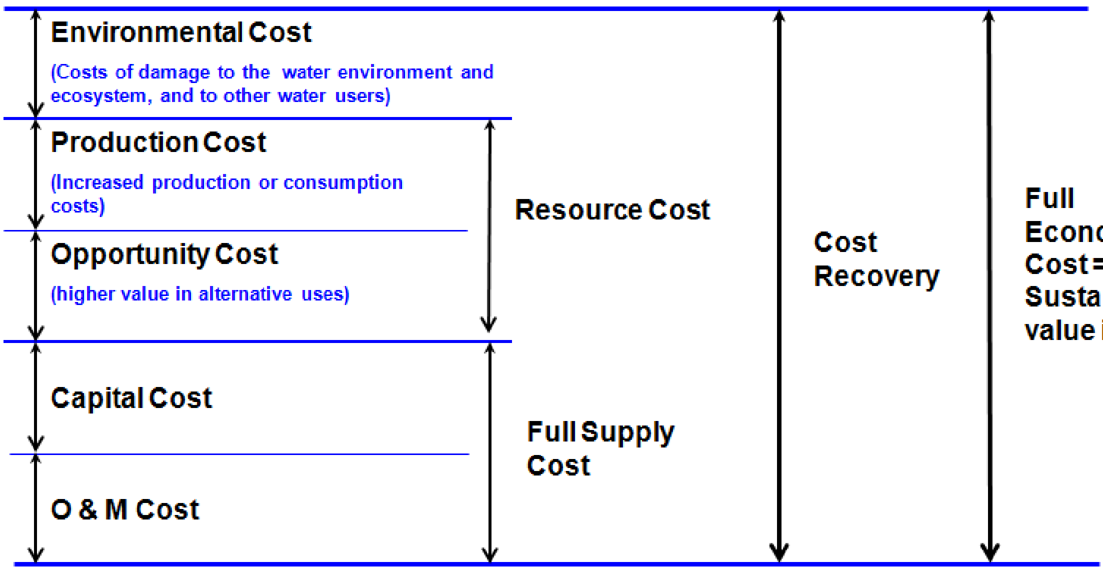
Figure 3.1 Definition of uses of water for the WFD (From: Moran and Dann, 2007)



Normally, water use values are defined in terms of changes in quantity and quality. This poses one of the main problems in their estimation, which is that similar uses in theory may have different impacts in practice; as an example; identical levels of in-stream abstractions may have different impact in quantity and quality depending on geographical and weather variations. Moran and Dann (2007) note that some texts of the Directive and related documents have used the term non-use to indicate non-abstractive uses (e.g. water for cooling purposes in hydropower and distilleries), as opposed to a passive activity totally unrelated to any direct exploitation. However, this is a disputed issue, even though water quality may not be altered as a result of a passive use, these types of abstraction have a temporal impact in water quantity. In light of the WFD, these types of use should also be included in the economic analysis.

The analysis of the full costs incurred by water users is also important under the WFD for the achievement of sustainable water pricing policies. The underlying aim is the rationalization of water use in Europe and the drive to increase efficiency of resource use, the argument goes that paying for the full costs of the service will dissuade the unsustainable over-use of water courses beyond their assimilative capacity. While the estimation of supply/discharge costs faced by some users may be quite straightforward, this includes the assessment of the financial costs faced by the water user. The assessment of the external costs associated with use is another story, much more difficult to depict in practice. In theory, the full cost estimation of water use should also include: the costs associated with damage to the water environment, associated costs caused to other users and the opportunity costs of use (related with the estimation of the higher value of water in alternative uses). Under the WFD, the internalization of the environmental costs and external economic and opportunity costs into prices for water services, named environmental and resource costs (respectively) in its legal text, is the ultimate aim of the cost recovery principle. The Directive specifically states that the application of the recovery cost principle should go beyond the simple estimation of the costs of supply of water services (capital, operation and maintenance costs) and mandates its application for the three major users of water services; industry, agriculture and households. Figure 3.2 introduces a schematic representation of our interpretation of the types of costs of water use under the WFD.

Figure 3.2 Typology of costs of water use under the WFD



A broad artificial distinction is made (for the sake of clarity in actual implementation) in the WFD legal text between the economic dimension (environmental and resource costs) and the financial dimension (capital and operation & maintenance costs) of the equation. Arguably, there are doubts about the ultimate theoretical value and practical application of the typology of costs proposed under the WFD (see figure 3.2). The figure could be read as though the full economic cost (of river restoration for example) is the sum of O&M costs, capital costs, resource, and environmental costs. It is often contested that these cost types are cost items that can simply be summed as they very often overlap. On the issue about categorising the external costs associated with water use for example, Brouwer (2006) further specifies opportunity costs as “the cost to society of use of the resource”, consisting of the direct economic user cost of water, the external cost that arises from water use, and the scarcity rent due to resource exploitation resulting in its non-availability for future use.

Ultimately and depending on the policy decision at stake, one can determine which cost categories are relevant to be included in an assessment. Specifically, a full cost-benefit analysis would take into account both financial and economic costs, while a financial cost-benefit analysis would not consider economic costs at all. Within a cost-effectiveness analysis, mainly financial costs are considered. The real difference between financial and economic costs lies of course in the question for whom the costs are assessed, the scope and the scale of the analysis.

Both financial and economic analyses have similar features. Both their goals are to estimate the net-benefits of a project investment against a baseline or counterfactual. But important distinctions worth highlighting for river restoration projects are (from WMO, 2007; Lago, 2008):

- The financial analyses compare benefits and costs exclusively relevant to the firm that is asked/encouraged to take some action, while the economic analyses goal is the pursuit of economic efficiency in public policy decision making and choosing options that are expected to deliver net benefits to society at large.
- In this respect while market prices are employed in financial analyses to assess investment decisions and ensure their financial sustainability, in economic analyses a conversion from the market price by excluding transfer payments4 (to assess all options net of tax and subsidies) is employed to derive economic prices.
- Therefore, financial and economic analyses also differ in their treatment of externalities, such as favorable effects on health or the environment. These are not covered in financial analysis.

In the context of river restoration projects, the application of a (financial) cost-effectiveness analysis is considered most relevant when it comes to the selection of alternative restoration measures with a given budget. For this purpose, the following sections will focus on the assessment and prediction of the financial costs related to river restoration projects.

3.4 Costs related to hydromorphological restoration

Costs take on many different characteristics, including the time frames during which they must be paid, the purposes (for direct costs) they serve, and the actors who pay them. As such, costs are best reported in a more complex manner than simply a single number. The categories of cost reporting are best informed by economic theory and a sensible breakdown for administrative reasons. For example, differentiating private costs for project implementation from opportunity costs borne by others can help provide a basis for a deeper analysis. A broad breakdown of conservation costs includes (Naidoo et al., 2006):

- acquisition costs,
- management costs,

[4] The UK Green Book (HM Treasury, 2013) on "the appraisal and evaluation in central government" defines a transfer payment as "*one for which no good or service is obtained in return. Social security payments are an example. They may change the distribution of income but they do not of themselves represent direct economic costs, except for any associated costs of administration or compliance. Transfer payments should be excluded from the costs and benefits in an appraisal*".

- transaction costs,
- damage costs, and
- opportunity costs.

Furthermore, the following costs must be considered specifically for restoration programmes:

- investment/construction costs.

In addition, there is the standard WFD-related cost typology which was developed for the CEA of the Programme of Measures. Article 4 of the WFD requires implementation of PoMs (including technical and policy instruments) to achieve environmental objectives (e.g., GES), which calls for a cost-effectiveness analysis. The breakdown of costs is represented below (see RPA, 2004):

- Non-recurring costs: these relate to capital costs but are one-off costs generated by a new measure/change in policy;
- Recurring costs: these include fixed costs (costs that do not vary across levels of production), variable costs (costs that vary across level of production or levels of activity) and semi-variable costs (costs that have both a fixed and a variable component);
- Non-recurring and recurring costs for regulators: these are associated with the set-up, administration and enforcement and monitoring of a new measure or a change in policy;
- Cost savings: these may arise from the adoption or implementation of a measure and include savings in materials (inputs), reduced energy requirements, the recovery or sale of by-product, reduced maintenance costs, reduced manpower requirements, etc.;
- Transfers: these are associated with taxes and subsidies. Financial costs to businesses will include transfer payments (implying that financial costs will differ from measures of economic cost);
- Non-water environmental costs/benefits resulting from implementing a measure: these include change in habitat, landscape, emissions to air, noise, etc. that may result from changes in land use (e.g. due to changes in agricultural practices or forestry), the construction of pumping stations and new water treatment plants, and other types of work, and
- Wider economic effects: any knock-on effects that are passed on or through to other sectors, organisations, etc. This includes the effects on producers and consumers in related market that are not captured by the estimation of direct non-recurring costs and recurring costs.

The costs of restoration projects are affected by many variables, some of which are project-specific, including weir height, and some of which are circumstantial, including regional variations in energy costs, labor costs, and requirements for monitoring and efficiency assessments. See for example Catalinas et al. (2014) who provide a detailed overview of methods and data used for cost estimation for freshwater habitat restoration planning under the WFD in Spain.

In general, the level of detail built into the cost typologies that are currently being used in functioning river restoration databases is not as high as outlined above. Looking more specifically at river restoration in Europe, cost typologies included in existing databases generally include only total project costs. An exception is the RESTORE database set up as part of the RESTORE LIFE+ project, whose cost typology includes total cost information for the following categories: investigation and design, stakeholder engagement and communication, works (i.e., construction works), post-project management, and monitoring. In most cases, however, information is only reported for "works."

Looking across the Atlantic, cost reporting in restoration databases does not seem to be appreciably more complex. The US National River Restoration Science Synthesis, as reported by Bernhardt et al. (2005), gathered cost data on thousands of projects implemented across the United States, but costs were only reported in terms of total project costs. Kondolf et al. (2007) worked with the same database alongside a set of interviews in California and pointed out the lack of useful project data for restoration projects, including cost data. The Utah Restoration Database is an example of a state-level database that also only reports costs at the project scale.

Within Deliverable 1.4 of REFORM, it was found that costs are reported in the literature mainly for the following three types measures from the measure typology considered for the REFORM: longitudinal connectivity improvement (through weir removal and fish passage installation), in-channel structure and substrate improvement (gravel cleaning or placement and installation of habitat diversification structures), and riparian zone improvement (revegetation).

Reviewing the previously mentioned typologies and factors that influence the costs of river restoration projects has led us to propose the following cost typology.

3.5 Recommended cost typology

The cost typology used for this analysis is based most closely on the one developed as part of the WFD cost-effectiveness analysis of PoM by RPA (2004). Specifically, we have adopted the non-recurring/recurring distinction in order to allow for insight into how costs develop over time. The list below illustrates the cost categories from the typology used in the database. These take account of the categories used in Deliverable 1.4 of REFORM.

- Non-recurring costs
- Planning and design costs
- Transaction costs
- Land acquisition costs
- Other construction / investment costs
- Recurring costs
- Annual maintenance costs
- Annual monitoring costs

Box 3.1 Definitions of relevant cost categories

Non-recurring costs are one-off costs that are also considered as fixed costs while **recurring costs** refer to regular cost incurred repeatedly, e.g. on an annual basis.
Planning and design costs include costs related to a variety of activities that are part of the preparatory project phase, including data collection, setting objectives, identifying outcomes, planning the schedule, identifying activities, developing the budget, selecting the project team, and setting up contingency plans.
Transaction costs may occur in the planning as well as in the implementation phase of the project and include communication charges, legal fees, informational costs, and quality control costs.
Land acquisition costs refer to the cost of the land/property that needs to be acquired for implementing the restoration project.
Other construction / investment costs refer to the cost of the factors which are needed to implement the project, including labour, material, equipment, financing, services, and utilities.
Maintenance costs refer to upkeeping and repair costs which occur over the duration of the restoration project; they are usually reported on an annual basis.
Monitoring costs occur after the restoration project has been implemented and refer to costs for labour and equipment that is needed to analyse the changes in ecological and hydromorphological conditions and the effectiveness of the measures implemented; they are usually reported on an annual basis.

The proposed cost-effectiveness analysis should be possible with the financial cost data covered by these variables. Cost-effectiveness analysis allows for a determination of which restoration projects should actually be pursued given a limited budget. Financial cost data, collected in the typology both as recurrent and non-recurrent costs, are combined with effectiveness or benefits data in order to establish a ratio or costs to benefits for each individual measure. The measures are then ranked according to their cost-effectiveness, and, if the target is known, summing the potential deployment of the most effective measures will reveal which of them should be implemented to reach the goal at least cost.

It should be noted that the evaluation of further cost categories (beyond financial costs) can be of importance to decision makers. A full cost-benefit analysis at the margin attempts to determine the efficient level of abatement either for one individual measure class or a basin as a whole. As such, the external economic costs of river restoration are needed to understand the full social costs of implementing these measures. By plotting the total social costs (derived from economic cost assessments) of measure implementation against the level of abatement, the relationship between the marginal costs of abatement (i.e., the costs of the next unit of abatement) and the level of abatement at that point can be derived (see Lago, 2008). By overlaying the marginal costs and marginal benefits of abatement action, the optimal amount of abatement can be found.

Interest rates, discount rates and depreciation are important variables in a proper financial cost assessment. The discount rate refers to the time value of money. In an investment decision, the discount rate serves as the multiplier that converts anticipated returns from a project to their current market value (present value). In the context of river restoration, the discount rate can be a determining factor when comparing alternative restoration measures. A high discount rate will reduce future costs and benefits while a low discount rate will increase them. In public-sector projects, the discount rates applied are usually in the range between 3.5 and 5.5 percent[5]. Depreciation, on the other hand, is a method of allocating the cost of a tangible asset (e.g. built infrastructure) over its useful life, i.e. over the duration of a river restoration project. In a proper costs assessment, the related costs are allocated, as depreciation expense, among the periods in which the asset is expected to be used.

Estimating non-recurring and recurring costs and taking into account discount rates and depreciation thus form the basis of a proper cost assessment. After the respective cost data have been gathered, they can inform a cost-effectiveness analysis and support the selection of alternative restoration measures. In order to provide some practical evidence, the following section will review examples where cost-effectiveness analyses have been carried out in order to select alternative measures in the context of river restoration projects.

3.6 Evidence and practical guidance for cost assessments in river restoration projects

As has been outlined in the previous chapters, a cost effectiveness analysis (CEA) could be a highly useful instrument for advising the selection of restoration measures. It enables decision-makers and stakeholders to compare project alternatives ex ante in order to assess which alternatives are financially viable. In addition, the role of CEA at the project level is twofold: i) to ensure that no other plan provides the same output for less cost; and ii) no other plan provides a higher output level for the same or less cost. Beechie et al. (2008) name cost effectiveness as one central approach in order to prioritize river restoration actions. The questions of interest here are how restoration measures or programs are selected in practice, what are the steps that lead to the selection of measures, what types of financial costs are normally considered and how these costs are estimated. In the following, we present practical examples from different river restoration practices focusing on understanding how cost effectiveness analyses have been used as a tool for selecting restoration measures. Subsequently, a set of guidelines on selecting cost-effective river restoration measures is deduced.

3.6.1 Examples of how restoration measures are selected in practice

In practice, many river restoration projects have not documented project costs (compare Bernhardt et al. 2005, chap. 1.b. of this paper). Instead of objective cost effectiveness criteria, the political context has been a considerable factor for the selection of restoration measures in the past. Kondolf et al. (2008) state that for a number of restoration programs in California, those measures were chosen which were politically visible and easily implemented. Moreover, “there was not a systematic, comparative process for prioritizing projects; prioritization was often based on ease of implementation

[5] The European Commission suggests a discount rate for public investments of 3.5% (and 5.5% for EU member states with Gross National Income below average) (EC, 2008) Furthermore, The UK and France current discount rates for public investments are 3,5% for the first 30 years of a programme and 4% respectively (HM Treasury, 2013 and Evans et al., 2006).

or intuitive appeal to agencies or stakeholders" (Kondolf et al. 2008, p. 936). The authors criticize that a decision for project measures based on such grounds does not necessarily lead to the most cost-effective outcome. Here, the use of a CEA could support the decision-making process and give unbiased information to politicians and project practitioners.

Box 3.2 Relevance of cost assessments in publicly available guidelines.

Within D6.3 of the REFORM project, a review of guidelines for the planning process and design of cost-effective and hydromorphologically relevant restoration and mitigation measures has been carried out.[6] According to the preliminary results of the review, few of the evaluated guidelines contained specific recommendations for conducting a cost-effectiveness analysis (CEA). The majority of the reviewed guidelines focussed on the hydromorphological and ecological components in the design of river restoration projects. Specific recommendations for the design of comprehensive cost assessments at project scale and the subsequent integration of cost estimates into a CEA seem to be largely underrepresented in publicly available guidelines. Comprehensive discussions on the relevance of cost assessments in the selection of alternative restoration/management measures can be found in Eberstaller (2007), DWA (2009) and Cramer (2012).

Some well-documented examples of river restoration projects for which CEA have been conducted can be found in the USA. In accordance with the Clean Water Act, restoration is seen as a tool to improve the chemical, physical and biological condition of impaired streams (EPA 1995). A legal document on the "Principles and Requirements for Federal Investments in Water Resources" establishes guidance for publicly funded river restoration projects. It states that the process of plan selection has to be fully reported and documented, and establishes cost effectiveness as one criterion to evaluate alternative measures (U.S. Council on Environmental Quality, 2013).

A great number of river restoration projects have been completed and reported in the USA.7 In the following we present two restoration plans in detail: first, the restoration of the Malden River in Massachusetts, and second the revitalization of the Los Angeles River in California. The two examples show how CEA have aided the decision-making process at different scales and exemplify the U.S. procedure for restoring rivers.[8]

Malden River, USA

A detailed project report was prepared for the restoration of the Malden River watershed, which is a sub-basin of the Mystic River of approximately 11 square miles and flows through the densely populated cities of Malden, Everett and Medford in Massachusetts, USA (see figure 3). The watershed is characterized by a low water quality as well as a low degree of biodiversity. Furthermore, an invasive plant species is dominating the

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[6] D6.3 Guidelines and decision support for cost-effective river-floodplain restoration and its benefits (scheduled October 2015).

[7] Compare database of U.S. Army Corps of Engineers

http://cw-environment.usace.army.mil/retro/index.cfm

[8] We focus on the two examples since beyond issues of scales, the use of more case studies would not have brought more value for our analysis.

riparian wetlands along the riverbanks. Primary objectives of the Malden River Restoration Project are the restoration of wetlands, of aquatic habitats as well as of fish migration (USACE 2008, p. ES-i).

For this rather small-scale project, a feasibility study was conducted by the United States Army Corps of Engineers (USACE), New England District. In preparation of the restoration actions, the study first identified environmental restoration needs and opportunities in the Malden River, next developed plans and cost estimates for various restoration alternatives, conducted a cost effectiveness analysis as well as an incremental cost analysis and finally, based on the results of the cost analysis, selected a recommended restoration plan (see USACE 2007). Five different restoration measures have been considered in the analysis: the removal of invasive species, removal of invasive species combined with the restoration of wetlands, creation of wetlands, placement of gravel or sand, and the establishment of a fish passage. The geographic area considered for restoration has been divided into six sub-areas. For each sub-area, the costs of the restoration measures to be applied were evaluated. Then, by means of the CEA it was determined which measures are cost effective in each of the sub-areas and furthermore which combination of measures across the sub-areas is the most cost effective one.

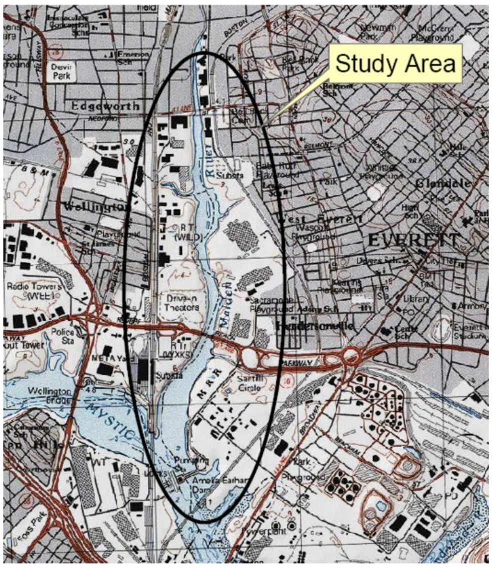


Figure 3.3 Topografic Map Malden River Restoration Project – Study Area (USACE 2008, p. ES-ii)

Costs were estimated by means of a cost estimation software of the USACE. Cost categories included in the estimation were costs for developing plans, real estate, engineering and design during construction, construction supervision, and a 20 percent contingency, as well as operation and maintenance costs over a 50-year project life. Costs have been analyzed in relation to the environmental output of the alternative, which is measured in Habitat Units (HU). The HU is based on the Habitat Suitability Index approach, which is then multiplied with the available area (for more detail, see USACE 2007, p. C-2, f).

Table 3.3 shows the total costs of each measure for different sub-areas and the expected number of habitat units created by each plan relative to the no action alternative. The fifth column indicates the costs per habitat unit, on which the selection of the final measures have been based, next to other considerations. The final plan included the following measures: remove invasive species & restore wetland in the sub-areas 3, 4, and 5; create wetland in sub-area 4; place gravel or sand in sub-areas 1 to 6 and establish the fish passage (G, H, I, K, L, M, O, P and Q).

Table 3.3 The case of the Malden River project: Costs and output of alternative plans (Adapted from: USACE 2007, p. C-11)

Plan ID	Restoration plan	Total cost ($000)	Habitat Units (HU)	Cost per HU ($000)
A	Total remove of invasive species in sub-area 2	792.7	0.54	1468.0
B	Total remove of invasive species in sub-area 3	1096.8	0.67	1637.0
C	Total remove of invasive species in sub-area 4	1443.9	1.02	1415.6
D	Total remove of invasive species in sub-area 5	1091.3	2.57	424.6
E	Total remove of invasive species in sub-area 6	8080.1	4.12	1961.2
F	Remove invasive species & restore wetland in sub-area 2	812.1	3.65	222.5
G	Remove invasive species & restore wetland in sub-area 3	1150.4	8.52	135.0
H	Remove invasive species & restore wetland in sub-area 4	1500.5	9.26	162.0
I	Remove invasive species & restore wetland in sub-area 5	1137.1	12.05	94.4
J	Remove invasive species & restore wetland in sub-area 6	8279.7	39.41	210.1
K	Create wetland in sub-area 4	1322.2	15.71	84.2
L	Place gravel / sand in sub-area 1	7.8	0.70	11.1
M	Place gravel / sand in sub-area 3	75.1	0.69	108.8
N	Place gravel / sand in sub-area 4	76.7	0.84	91.3
O	Place gravel / sand in sub-area 5	48.7	0.42	116.0
P	Place gravel / sand in sub-area 6	84.1	0.79	106.5
Q	Fish passage	716.4	49.04	14.6

Next, the incremental cost analysis “examines how the costs of additional units of environmental output increase as the level of environmental output increases” (USACE 2007, p. C-5). The cost analysis for the Malden River, comprised of a CEA and incremental cost analysis (ICA)9, thus enabled identifying those restoration plans that are most cost effective in providing environmental benefits (outputs), eliminating inefficient plans, and determining if plans are cost effective. Furthermore, “the analysis aids decision making by ensuring that the least cost solution (“Best Buy Plan”) is identified for all possible levels of environmental outputs” (USACE 2007, p. C-1).

Los Angeles River, USA

[9] Incremental cost analysis is a tool for plan formulation and evaluation which examines “how the costs of additional units of environmental output increase as the level of environmental output increases” (USACE 2007, p. C-5).

An approach quite similar to the Malden River project was chosen for the Los Angeles River Revitalization. The 51-mile-long Los Angeles River is the central stream of an 870 square mile watershed located in Southern California (see figure 3.4). It flows through the second-largest urban area of the U.S.A. and can be characterized as heavily modified: Most of the river has been encased in concrete banks as well as concrete bed, its channel has been widened and deepened and the river's course been straightened. This resulted in the disconnection of the river from its floodplain as well as from other ecologic zones, and a low degree of biodiversity. For the restoration, a study area of eleven miles was chosen which has the highest potential for restoration (e.g. large area without concrete river bed). Objectives of the restoration project are to restore ecological processes and biological diversity, to increase habitat connectivity and to increase opportunities for recreation (USACE 2013b).

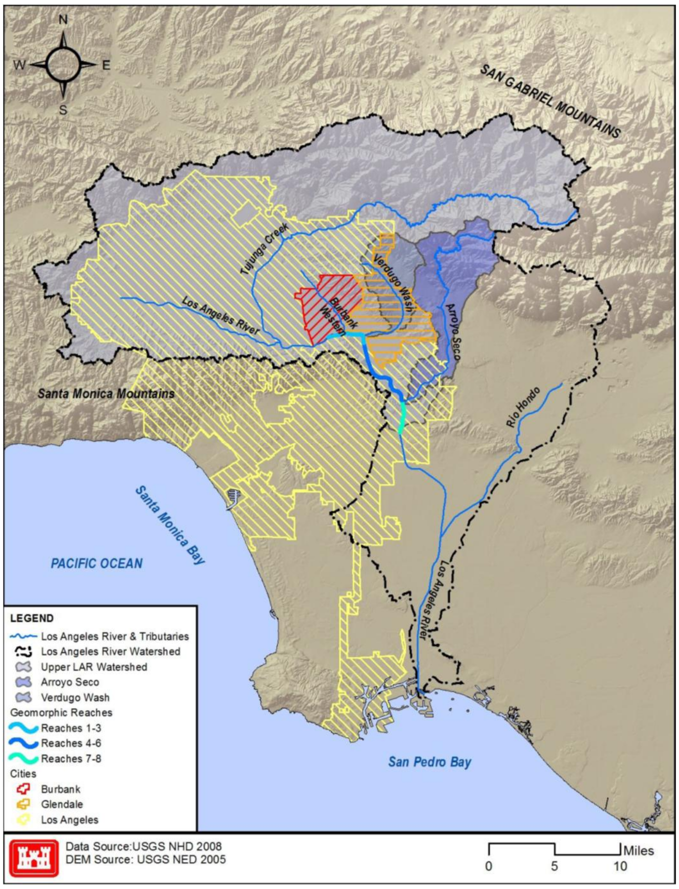


Figure 3.4 Los Angeles River Watershed (USACE 2013b, p. 3-2)

In the first stage of the project, management measures were developed by a large team of experts under consideration of the ecological conditions, constraints to the project, different land-uses along the river and the practicability of implementation. Consequently, 19 alternative plans were formulated (compare USACE 2013b, p. 4-21, f), depicting different combinations of measures. Moreover, the study area was divided into sub-areas according to geo-morphological sections of the study area. The 19 alternatives were applied and adopted to the conditions of each of the sub-areas. Next, a cost evaluation – comprised of a CEA and an ICA – was conducted, using a software-based evaluation tool. The following types of costs were included in the analysis (USACE 2013a):

- Construction costs
- A contingency for construction of 25%
- Real estate costs, developed for each alternative and each sub-area
- Relocation costs, e.g. for businesses that would require relocation
- Mobilization and demobilization costs, which includes transporting equipment and crews to the project site, setting up site facilities and staging areas. Mobilization and demobilization costs were estimated to be 7.5% of construction costs.
- Planning, engineering and design costs, which cover the preparation of plans, specifications as well as engineering during construction. These costs were estimated at 11% of construction costs.
- Supervision and administration costs cover the construction management during construction. These costs were estimated to be 6.5% of construction costs.
- Operation and maintenance costs, which are defined as costs for the routine work that is expected to occur each year over the 50-year life cycle of the project. Operation and maintenance costs were estimated by using percentages of the original installation cost for individual items (e.g. concrete demolition).

One principle of the L.A. River restoration project was that land acquisition costs should be kept to a minimum. Restoration plans for which land acquisition costs exceeded 25% of the total costs were eliminated (cf. USACE 2013b, p. xxiv).

As in the Malden River project, effects were not regarded in monetary terms. Habitat benefits were measured in habitat units, which were examined in relation to the project costs in order to “ensure cost-effective and justified plans” (USACE 2013b, p. 4-34). Furthermore, the software enabled a recombination of individual measures and sub-area plans in the economic analysis.

By means of the CEA, for each alternative plan and each sub-area it has been scrutinized whether the expected output could be achieved more cost-effectively by another alternative. Some measures were eliminated during the planning process since they have been found to be ineffective, inefficient, incomplete or unacceptable. The CEA was one instrument to phase out inefficient measures, e.g. the considered creation of underground basins for attenuation. The measure would have provided little to no additional value for the associated costs, thus the measure was dropped. Another example is the proposed plan to construct tunnels or large culverts to divert storm season flows around the project reach. For this measure, the cost analysis showed that the costs of construction and land acquisition would have largely exceeded the expected benefits. Therefore, this measure was also excluded from further consideration (compare USACE 2013b, p. 4-10, f).

The CEA resulted in a subset of cost-effective plans. Subsequently, the ICA examined the sequential increase of outputs in order to determine whether increasing levels of restoration are worth the added cost. Based on this, the most cost-effective plans, i.e. those plans with the lowest incremental cost per unit of output, could be ranked and selected. Based on the results of the cost analysis and under consideration of the study objectives, a final array of four alternative plans was chosen. All of them were considered as cost effective and in line with the restoration targets.

- "Alternative 10 […] provides restoration in all sub-areas, restores habitat at the Piggyback Yard, includes some widening at Taylor Yard, and provides transitions or connections between existing riparian corridors and concrete lined river reaches.
- Alternative 13 […] includes all the features in Alternative 10, and adds additional restoration in the natural bed reaches of the Glendale Narrows, increased widening in Taylor Yard, and restoration at the Arroyo Seco confluence.
- Alternative 16 […] includes all the features in Alternatives 10 and 13, and widens and adds terracing in Reach 5, and adds terracing, widening, concrete removal in the channel bed, and restored wetlands in the channel and in Piggyback Yard.
- Alternative 20 […] includes all the features of Alternatives 10, 13 and 16, and adds widening in Reach 2, restores the confluence with Verdugo Wash in Reach 3, and restores wetlands at the Los Angeles Historic 45 State Park with a terraced connection to the mainstem in Reach 7" (USACE 2013b, p. 5-1).

After scrutinizing the final four alternatives in more detail, the report recommends the realization of alternative 13, as it best meets the restoration objectives at the lowest relative costs. Alternatives 16 and 20 would reach higher ecological benefits, but with a higher relative increase in cost (see table 3.4). Alternative 10 only minimally fulfils the restoration targets. This recommendation led to a broad public discussion by the local community. Advocacy groups and elected officials led by the city's mayor started a campaign for alternative 20, which offers higher ecological and recreational benefits. Campaigning activities included lobbying at the White House and relevant authorities. In the end, public pressure led to the adoption of alternative 20 .10

Table 3.4 Final Array cost comparison L.A. River restoration (Adapted from: USACE 2013, p. 6-44)

	Alternative 10	Alternative 13	Alternative 16	Alternative 20
Total First Cost	$ 374,782,639	$ 453,406,057	$ 803,928,734	$ 1,080,627,338
Incremental First Cost	$ 374,782,639	$ 78,623,418	$ 350,522,677	$ 276,698,604
Incremental First Cost %		21%	77%	34%
Average Annual Habitat Units (AAHU)	5,321	5,902	6,509	6,782
Incremental AAHU		581	607	273
Incremental AAHU %		11%	10%	4%

[10] Los Angeles Times article of 28th of May 2014, http://www.latimes.com/science/la-me-la-river-approval-20140529-story.html

	Alternative 10	Alternative 13	Alternative 16	Alternative 20
Total Cost/AAHU	$ 70,435	$ 76,822	$ 123,510	$ 159,338
Total Cost/AAHU % Increase		9%	61%	29%

Comparative analysis and conclusions

Comparison of the Malden River and L.A. River projects shows that a similar process led to the final choice of restoration measures, with a CEA playing a central role in the preparation of the decision-making. The steps towards selecting restoration measures were the following: First, the environmental restoration needs and opportunities were identified. This was followed by the development of restoration alternatives. Next, cost estimates for the various alternatives were made and a CEA as well as ICA was conducted. Finally, restoration measures were selected.

The similar approach of the two projects is not a coincidence. Both projects are led by the U.S. Army Corps of Engineers, whose restoration activities are guided by internal guidelines and policies. For example, a manual on the evaluation of environmental investments regulates that a CEA as well as an ICA need to be performed (Robinson et al., 1995). The manual, in turn, has been based on the U.S. Water Resources Council's "Principles and Guidelines for Water and Related Resources" of 1983, which have been revised in 2013 (U.S. Council on Environmental Quality, 2013). The document establishes a common framework for water related federal investments, it establishes project selection criteria and prescribes how to calculate costs and benefits. More precisely, river restoration planners are obliged to consider and evaluate a range of reasonable alternatives and the selection of measures has to be justified by comparing the benefits to the costs. Thus, these principles establish a standardized procedure for planning river restoration activities.

Regarding the restoration measures in the Malden and L.A. River projects, the different volume and scope of the projects becomes obvious. While for the Malden project, the considered measures are relatively limited as shown in table 3, for the L.A. project a very wide range of restoration actions was examined in preparation of the restoration (cf. USACE 2013b, pp. 4-12 ff.). Some examples include concrete removal, widening of tributary channels, daylighting underground pipes, planting and constructing a bridge undercrossing for wildlife.11

The considered cost categories are rather similar in the two examples. In both, construction costs, costs for planning, engineering and design, real estate costs, supervision costs as well as operation and maintenance costs were covered in the analysis. One difference is that in the L.A. project relocation costs as well as mobilization costs for the transport of crew and equipment are included in the cost estimation, while these cost types are not considered in the Malden River project plan. In comparison to the recommended cost typology of this document (cf. chapter 2.3), it can be stated that the considered cost types in the U.S. examples are in large part captured in the proposed

[11] Many of those measures and their purposes have been described in more detail in Deliverable 1.4 of REFORM, chapter 3.3.

categories. Regarding the non-recurring costs, planning and design costs as well as land acquisition costs ("real estate costs") were considered in the U.S. projects. Further cost types of the examples, such as the supervision and administration costs, can be assigned to the proposed category "other construction / investment costs". Transaction costs are not explicitly listed in the two project plans. In regard of the recurring cost, only annual maintenance costs were considered in the L.A. and Malden projects, while monitoring costs are not mentioned in the respective cost estimations. In summary, the here promoted cost categories appear to be useful for the practical realization of CEAs in preparation of river restoration projects.

In both of the examples described in this chapter, a CEA played a central role in the process of selecting restoration measures. The CEA aided in prioritizing restoration measures and plans, developing cost effective combinations of measures and eliminating cost ineffective plans. Moreover, the examples show that a CEA is a useful tool for both small-scale and large-scale restoration projects and applicable under consideration of a very different range of restoration measures. However, the most cost effective measures do not necessarily have to be adopted, as cost effectiveness is neither a binding, nor is it the only decisive factor to be considered. This could be seen in the L.A. River restoration process, where the cost analysis suggested Alternative 13 be the optimal combination of measures (USACE 2013b). However, in consequence of a campaign by the city for the more ambitious restoration plan, Alternative 20 was preferred by the decision-makers (compare case study section above on Los Angeles River). Thus, the CEA does not prescribe the measures which have to be taken; rather its ultimate goal is to help making informed and sound decisions. Moreover, in the U.S. examples the budget was yet undecided during the planning phase. Therefore, the CEA aids in determining the size of the budget and volume of the project. In contrast, in river restoration projects in the EU, often the budget for the project is granted first, then the CEA is used to decide on which measures are to be realized (e.g. Spain, see Deliverable 1.4 of REFORM).

3.7 Practical guidance for identifying cost-effective restoration measures

This section will present a quick reference guide for identifying cost-effective river restoration measures at project level. The guidance is based on the "Evaluation of Environmental Investments Procedures Manual" designed by the U.S. Army Corps of Engineers (Robinson et al., 1995) and complemented with the cost assessments recommendations outlined in the previous sections. In the following, the individual steps analysing the cost-effectiveness of alternative river restoration measures are outlined:

A. Plan formulation steps

Plan formulation is about generating all possible alternative plans from the management measures under consideration. Within this step, a comprehensive list of feasible ecological restoration techniques is established. Instream, riparian, and upland techniques should be considered, individually and in combination. For these, outputs and costs should be assessed according to the following procedure:

1. Display Outputs and Costs of Management Measures
2. Identify Management Measure Relationships
3. Add Costs and Outputs of Combinations

B. Cost-Effectiveness Analysis Steps

The actual cost-effectiveness analysis is then carried out for alternative measures or combinations of measures. A selected restoration technique should be cost-effective, in addition to resulting in major environmental benefits. Thus, economic criteria (i.e. relevant cost categories) are part of the technical process to determine whether restoration techniques are reasonable. The following steps should be carried out:

4. Identify and select "Production Effective" (cost effective) Solutions
5. Identify and reject "Production Ineffective" (non-cost effective Solutions

C. Incremental Cost Analysis Steps

Within the incremental cost analysis, the incremental cost incurred and incremental output provided as the project scale is increased, is calculated and displayed. This analysis is based on the cost-effective measures or combinations of measures identified in Step 5. The information provided by Step 6 is intended to support the selection of the appropriate project scale

6. Calculate and Display Incremental Costs

D. Additional Analytical Steps to Assist in Scale Selection

Additional analytical steps can be carried out in order to assist in selecting the appropriate project scale (Step 6). Specifically, the following steps help 'smoothing' fluctuations in incremental unit cost and further illuminate rises in incremental unit costs:

7. Calculate Change in Unit Cost from No-Action Plan to All Other Plans
8. Recalculate Change in Unit Cost from Last Selected Plan
9. Tabulate and Display Incremental Costs of Selected Plans

4 The benefits of river restoration projects

River restoration provides a wide array of hydrological, ecological and socio-economic benefits. Many of these benefits are so-called public goods and services provided by restored or natural river systems, and can only be estimated in monetary terms using non-market valuation techniques. A limited number of such non-market valuation studies exists, which are summarized and synthesized in a structured way in a meta-analysis in this chapter. The chapter consists of the following parts: first the data collection procedure is described in section 4.1., followed by a presentation of the created variables in section 4.2. The design of the database is described in 4.3, including the presentation of the summary statistics of the variables. The econometric model underlying the meta-regression analysis is presented in section 4.4. and is followed by a discussion of the results of the estimated model in section 4.5. Section 4.6 concludes by focusing on the reliability of the estimated meta-regression model for predicting the non-market values of river restoration using benefits transfer.

4.1 Data collection

Potential articles about the socio-economic benefits of river restoration were selected based on two criteria. First, the articles were required to address river restoration. The *REFORM restoration measure typology* in Ayres et al. (2014) was used as a guideline to determine whether the measures evaluated in a particular study could be seen as river restoration measures. Second, in order to be selected, the article had to focus on the economic valuation of the impacts of the river restoration measures analyzed in a study. The studies included in the database are listed in Table 4.1.

The relevant scientific articles were searched via Google Scholar and the e-library of the VU University Amsterdam (http://elibrary.vu.nl/). In the search process we used key words such as *river*, *stream* and *watershed* to indicate the relevant type of waterbody. The words *restoration*, *rehabilitation* and *instream flow protection* were used to indicate the relevant type of improvement to be valued. *Contingent valuation*, *choice experiment*, *willingness to pay* and *willingness to accept*, and their abbreviations *WTP* and *WTA* respectively, were used to search for relevant non-market valuation methods.

The data provided in the collected papers were complemented with publicly available economic and socio-demographic data, climatic and geographic characteristics of the river study locations, and information derived from maps and related river images available on the web.

Table 4.1. List of articles included in the database and number of value estimates per study

Nr	Authors	Journal[a]	Study year	Nobs[b]
1	Hanley et al. (2006)	ERAE	2005	9
2	Bliem et al. (2012)	JEM	2007&2008	9
3	Bliem and Getzner (2012)	EEPS	2007&2008	6
4	Grossmann (2012)	EE	2010	1
5	Grossmann and Dietrich (2012)	WRM	2008	1
6	Hanley et al. (2006)	JEM	2001	9
7	Nardini and Pavan (2012)	JFRM	2004	1
8	Paulrut and Laitila (2013)	AE	2008	3

Nr	Authors	Journal[a]	Study year	Nobs[b]
9	Jørgensen et al. (2012)	EE	2009	4
10	Ramajo-Hernández and Saz-Salazar (2012)	ESP	2010	1
11	Stithou et al. (2012)	TESR	2010	4
12	Soliño et al. (2013)	IJER	2007	6
13	Saz-Salazar et al. (2009)	STE	2006	5
14	Gómez et al. (2014)	JH	2014	1
15	Grazhdani (2013)	IJIRSET	2012	1
16	Honey-Rosés et al. (2013)	EE	2012	2
17	Perni et al. (2011)	WEJ	2009	3
17	Meyerhoff and Dehnhardt (2007)	EuroE	2005	2
18	Acuña et al. (2013)	JAE	2008	1
20	Alam (2008)	IJWR	2001	1
21	Alam (2013)	JDA	2001	1
22	Han et al. (2008)	EIAR	2002	1
23	Kenney et al. (2012)	JAWR	2008	4
24	Holmes et al. (2004)	EE	2002	4
25	Zhao et al. (2013)	STE	2008	3
26	Loomis et al. (2000)	EE	1998	1
27	Weber and Stewart (2008)	RE	2006	8
28	Qiu et al. (2006)	JAWR	2002	2
29	Meyer (2013)	ERE	2008	2
30	Ojeda et al. (2008)	EE	2006	2
31	Berrens et al. (1998)	EE	1995	1
32	Che et al. (2014)	EM	2012	6
33	Collins et al. (2005)	WRR	2003	6
34	González-Cabán and Loomis (1997)	EE	1995	2
35	Lee (2012)	WI	2009	3
36	Zhao et al. (2013)	ERE	2008	6
37	Zhongmin et al. (2003)	EE	2001	3
38	Schultz and Soliz (2007)	JAWR	2007	2
39	Tunestall et al. (1999)	JEPM	1995&1997	2

[a] Abbreviations: *AE* Applied Economics; *EE* Ecological Economics; *EEPS* Environmental Economics and policy Studies; *EIAR* Environmental Impact Assessment Review; *EM* Environmental Management; *ERAE* European Review of Agricultural Economics; *ERE* Environmental and Resource Economics; *ESP* Environmental Science and Policy; *ESR* The Economic and Social Review; *EuroE* European Environment; *IJER* International Journal of Environmental Resources; *IJIRSET* International Journal of Innovative Research in Science, Engineering and Technology; *IJWR* International Journal of Water Resources; *JAE* Journal of Applied Ecology; *JAWR* Journal of the American Water Resources; *JDA* The Journal of Developing Areas; *JEM* Journal of Environmental Management; *JEPM* Journal of Environmental Planning and Management; *JFRM* Journal of Flood Risk Management; *JH* Journal of Hydrology; *RE* Restoration Ecology; *STE* Science of the Total Environment; *WEJ* Water and Environmental Journal; *WI* Water International; *WRM* Water Resource Management; *WRR* Water Resources Research;
[b] Number of observations in each article.

4.2 Database variables

4.2.1 Variable categories

Variables in the database are divided into ten different categories, which in some cases have sub-categories as well. *Database management* (1) assigns an identification code (ID) to studies and estimates included in the database. *Bibliographics* (2) provides author names, titles of articles, publication year, type of publication, and journal names, together with volumes and pages. *Study typology* (3) states which non-market valuation techniques were used, types of survey administration, elicitation formats (for contingent valuation studies), number of cards (for choice experiments), statistical models, and survey or study year. The category *Restoration measures* (4) is divided into nine classes with measure descriptions, as well as a variable for the estimated costs of restoration. *Valuated attributes* (5) is divided into eight categories according to the attributes used in the choice experiments in the various studies, with descriptions of the attributes in each category. *Value* (6) provides details about the welfare measures reported in the articles, including their standard deviation, monetary units, inflation- and purchasing power parity (PPP)-adjusted values, payment vehicle etc. *Target population* (7) provides information about the population of beneficiaries in each study (the respondents), sample size, distances from water bodies, and income per household. *Ecosystem services* (8) divides the valued benefits into provisional, regulating, cultural, and supporting ecosystem services, and provides specifications for each ecosystem service. *Topographical details* (9) presents information about the region, country, country region, and city or municipality in which the restored rivers are located. Finally, *Riverbasin characteristics* (10) details the river basins, such as their names, catchment size, average annual precipitation and population density.

4.2.2 Description of selected variables

Many of the variables in the database are straightforward, such as river location and catchment size, and do not require additional explanation, while others have to be specified in more detail to ensure proper understanding of the database. Below we provide brief descriptions for some of these variables (see also Table 4.2).

Year of survey

Not every article provided information on when the survey or study was conducted, also not after contacting the authors. In one or two cases, the year the article was submitted to the journal was used as a proxy for the study year.

Restoration measures

Categorization of the proposed restoration measures was done according to the *Measure typology* outlined in REFORM Deliverable 1.4. This typology consists of nine classes of restoration measures, with multiple measures per class. In the database, each measure is shortly described and a measure code is assigned.

Attributes

Attributes is a variable category which was initially included for articles that report choice experiment (CE) results, and then extended to cover also contingent valuation (CV) studies. The selection of attribute categories was done based on the attributes used in the articles. Only restoration-related attributes (i.e. attributes describing and quantifying improvements resulting from the proposed measures) were included in the variable category *Attributes*. Other attributes, such as proximity or costs were included in other parts of the database.

Purchasing power parities

Purchasing power parities (PPP) have been used to adjust the monetary values, i.e. willingness to pay (WTP) and household income. The OECD StatExtracts website was used as a source for the PPP data (see OECD PPP (2014)). The PPP-adjustment of monetary values follows the following procedure:

$$x_{ad} = \left(\frac{x_{un}}{PPP_{loc}}\right) \cdot PPP_{EU}$$

where x_{ad} is the adjusted monetary value, x_{un} is the 'unadjusted' monetary value, PPP_{loc} is the PPP value of the specific country in which the survey or study took place and PPP_{EU} is the PPP value of the euro area. Both PPP's are for the year 2013 and therefore the unadjusted monetary value has to be corrected for inflation to the year 2013, before it is used as input into this formula.

Inflation correction

Annual consumer inflation indices were used to adjust the monetary values for inflation and make them comparable for the same 2013 price level. The OECD (2014) Consumer Price Index (CPI) was used as the main data source, and the inflation adjustment was as follows:

$$x_{2013} = x_t \cdot (1 + i_{t+1}) \cdot (1 + i_{t+2}) \ldots \cdot (1 + i_{2012}) \cdot (1 + i_{2013})$$

where x_{2013} is the inflation-adjusted monetary value, x_t is the unadjusted monetary value in year *t* and all *i*'s are annual inflation indices for years *t+1* to 2013.

Table 4.2. Description of selected variables in the database

Variable category	Variable	Short description
Valuation study	Elicitation format	In case of contingent valuation: Open ended (OE), payment card (PC) or dichotomous choice (DC)
	Number of cards	In case of choice experiment
	Statistical model	Abbreviations of statistical models such as RPL (random parameter logit) and MNL (multi nominal logit)
Restoration measures	Name of measure class	Dummy variables; 1 if proposed measure belongs to this class, 0 otherwise
	Measure description	Description of measure in wording from the article
	FORECASTER Code	Measure code, from FORECASTER measure typology, for specific measure type
Valued attributes	Name of attribute type	Dummy variables; 1 if attribute type was used, 0 otherwise
	Attribute units	Description of units of attribute
Value	Compensating/equivalent surplus	CS for compensating surplus measure, ES for equivalent surplus
	Description baseline	Short description of baseline scenario in wording of article
	Description policy scenario	Short description of policy scenario in wording of article
	Most drastic scenario	Dummy variable; 1 if valued scenario is most drastic scenario, 0 otherwise
	Monetary value 2013	Inflation adjusted value to the year 2013
	Measurement unit	Description of measurement unit such as 'per household per year'

Variable category	Variable	Short description
	Annual payment or shorter time interval	Dummy variable; 1 if proposed payment is annually, 0 if time interval between payments is shorter
	WTP for restoration policy or per attribute	Dummy variable; 1 if WTP is estimated per attribute, 0 if WTP is estimated for whole policy
	WTP for:	Description of attribute in case of WTP per attribute
	Standard deviation of WTP	Statistical spread oft he WTP value
	Adjusted standard deviation of WTP	Inflation and purchasing power adjusted standard deviation of WTP in order to allow for comparison
	PPP	Purchasing power parity
Target population	Average income per year per household	Net disposable income where possible
	Average income 2013 value	Inflation adjusted average income to the year 2013
	Monetary unit	Monetary unit of average income such as euro (EUR) or dollar (USD)
	Average income PPP (Euro area) adjusted	Inflation and purchasing power adjusted average income; adjusted to purchasing power in Euro area as a whole
	Net/Gross	Specification whether income is net or gross
	Average income of:	Specification of to whom the average income value relates; e.g. sample, country, etc.
	Mean distance from water body	Mean distance from water body for the specific respondents
	User/non-user	Whether or not respondents are users
Benefits	Name of benefit providing ecosystem service	Dummy variable; 1 if provided ecosystem service will improve/occur and provide additional benefits
	Specification	Specification of provided additional benefits in wording of article
River basin characteristics	Catchment size	Catchment size in squared kilometers
	Studied fraction of river	Fraction (value between 0 and 1), of river at which is focused in article, to indicate how the studied part relates to total river size
	River type	Description of type of river
	REFORM River type No	Number for class of river (1-8 for eight different river types); from Table 3 in Deliverable 1.4 (REFORM)
	Population density	Number of people per square kilometer
	Population density in:	Specification of area for which population density value was determined
	Climate	Description of climate type
	Climate code (Koppen-Geiger)	Koppen-Geiger climate classification code
	Mean annual precipitation	Annual precipitation in millimeters per year
	Mean temperature	Mean temperature in degrees Celsius
	Precipitation and temperature in:	Specification of location of which mean temperature and precipitation values were used

Sample size

We report, if possible, only the number of respondents who actually took part in a stated preference (choice experiment or contingent valuation) study. That is, if the total number of respondents approached during a particular study was *N*, but response rate was only *z%*, then *z*N* was reported as the actual sample size.

Ecosystem services

Categorization of ecosystem services types included in the database was done based on the benefits described in the articles. Benefit types were divided into provisional, regulating, cultural and supporting ecosystem services.

River type classification

River type classification is based on at least two of the following criteria: satellite images of the area, photos along the river course, approximations of its gradient, and information on dominant alluvial material (sand, grain, silt, etc.). The types are determined according to the specifications provided in Table 3 of REFORM Deliverable 1.4 (see Ayres et al (2014)). Satellite images from Google Maps were used to determine if a particular river has a single thread or multiple threads. Figure 4.1 gives examples of a single thread and a multiple thread river.

Figure 4.1. Examples of a single-thread river (Clyde, UK) and a multiple-thread river (South Platte, USA) Source: (maps.google.com)

Information on dominant alluvial material was searched for on the internet. Photo images were used to determine dominant alluvial material if the definite answer could not be found in other information sources. Figure 4.2 provides an example of such an image that was used to classify one of the rivers.

Figure 4.2. Example of a non-map image used to determine river type (Chiese River, Italy, alluvial gravel bed) Source: (Panoramio.com)

Climate variables

The Koppen-Geiger climate classification in combination with maps by Peel et al. (2007) were used to code the climatic condition of river restoration sites. The classification code was based on the color coding of the river locations in the map. Average annual precipitation and average temperature were taken for a city located near the studied river. Climate Data (2014) and Weatherbase (2014) were used as sources for European and non-European rivers respectively.

4.3 The database

The database contains 39 different scientific articles that assess the non-market value of river restoration projects, as presented in Table 4.1. The surveys or studies presented and discussed in these articles were conducted within a time span of 18 years, between 1995 and 2013, although only four studies were conducted before 2001, see Figure 4.3. Geographically, the majority of studies come from Europe (22 papers), followed by America (12 papers) and Asia (5 papers), see Figures 4.4 and 4.5.

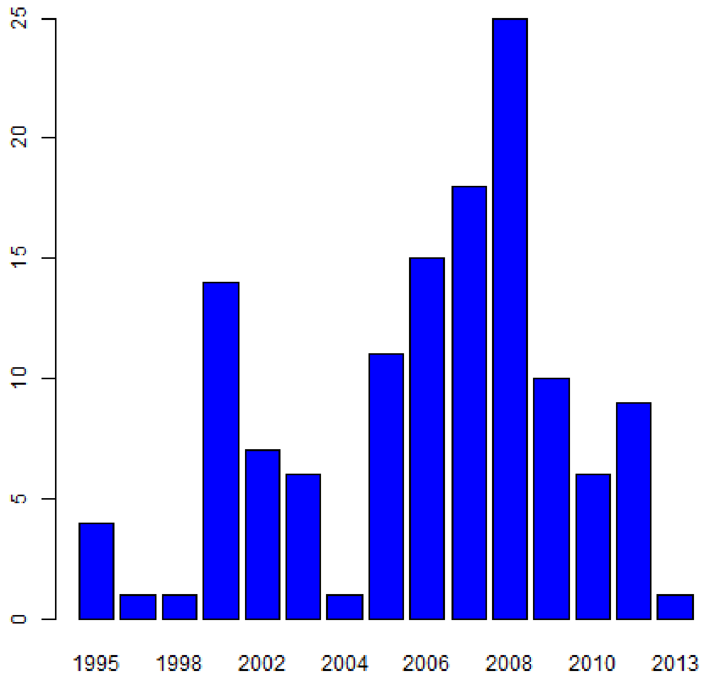


Figure 4.3. Distribution of number of papers in the database by year

Overall, 129 monetary values were extracted from those papers, including 88 mean WTP and 24 marginal WTP estimates, adjusted for PPP and inflation and expressed in 2013 price level euros. In terms of valuation methods, contingent valuation (CV) is used as valuation technique in 21 of these articles, choice experiments (CE) in 11 articles, and in the rest other non-market valuation techniques are used. For the meta-analysis, we limit our database to those papers focusing on CE and CV estimates only. This gives us 29 papers with 109 monetary observation, including 86 mean WTP estimates (see the summary in Table 4.3).

As noted before, in addition to the monetary welfare variables, the database contains several groups of variables that describe the papers included in the database and provide details of the study designs, ecosystem services under valuation, the river restoration measures, details of the geographical location of the river sites and river basin characteristics, and demographic characteristics of the respondents targeted in the surveys.

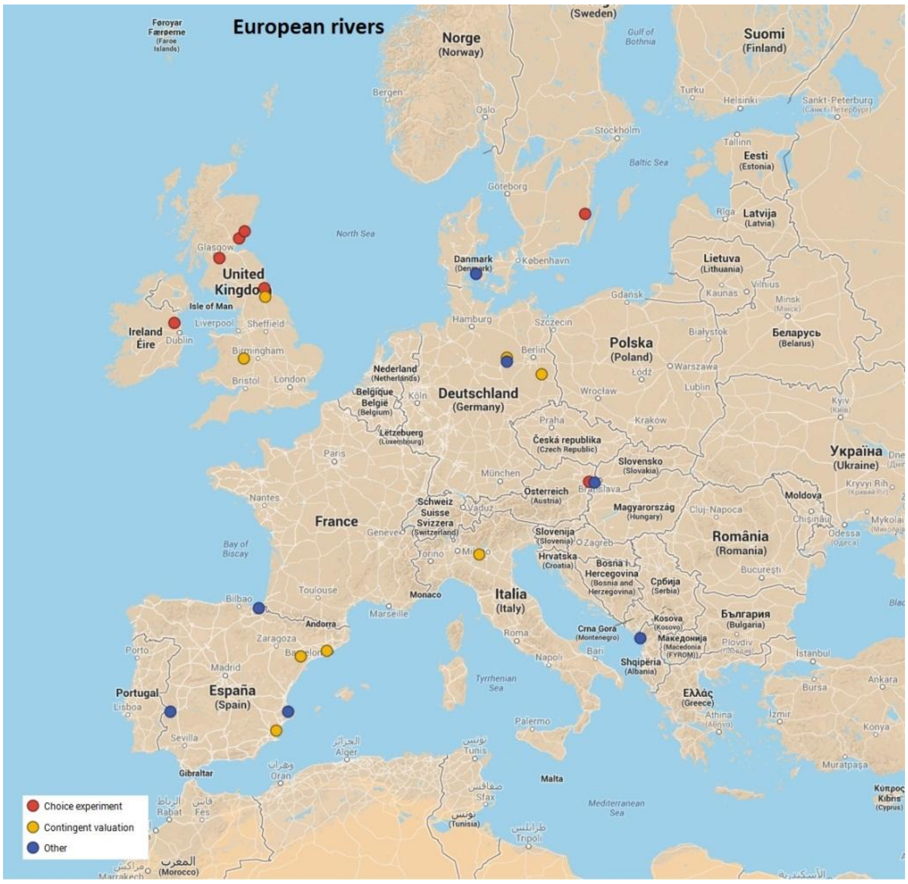


Figure 4.4. Locations of the river restoration studies in Europe

Figure 4.5. Locations of the river restoration studies outside of Europe

Table 4.3. Summary of the number of estimates across value elicitation formats

Study formats	# of WTP values	# of marginal WTP values
Choice Experiment	35	21
Contingent Valuation:		
dichotomous choice	21	
polychotomous choice	25	1
open-ended format	5	1
Total:	86	23

Restoration measures

Many of the proposed restoration measures could not be categorized into one of the eight classes and had to be classified as 'other measures' (the ninth class of measures). This becomes clear by looking at the last row of Table 4.4, which shows that more than half of the articles proposed at least one measure which had to be categorized as 'other measure'. The distribution of restoration measure frequencies is very uneven: riparian zone improvements were considered in almost 36% of the articles, while sediment flow quantity improving measures proved to be the least frequently studied class of measures in river restoration and included in only one article.

Table 4.4. Distribution of articles and estimates across river restoration measures

FORECASTER measure class	Class code	Number of articles	Number of estimates
Water flow quantity improvement	01	6	19
Sediment flow quantity improvement	02	1	4
Flow dynamics improvement	03	4	14
Longitudinal connectivity improvement	04	3	10
River bed depth and width variation improvement	05	5	15
In-channel structure and substrate improvement	06	6	18
Riparian zone improvement	07	14	52
Floodplains/off-channel/lateral connectivity habitats improvement	08	7	30
Other hydrological / morphological improvements	09	20	78

River types

The results of grouping the collected articles and derived monetary estimates according to the REFORM river types is shown in Table 4.5. Similarly to the grouping of river restoration measures, the distribution of articles and estimates is very uneven across the river types. The majority of studies were performed for one-thread rivers with sand or gravel bedrocks, while multi-thread rivers are underrepresented in the database.

Table 4.5. Distribution of articles and monetary estimates across river types

REFORM river types	Number of articles	Number of WTP estimates
1-thread bedrock	1	5
1-thread coarse beds	2	8
1-thread on gravel	5	21
m-thread on gravel	0	0
1-thread on sand	19	60
m-thread on sand	4	6
1-thread on silts	1	5
m-thread on silts	1	0
multiple rivers	6	24

Valuation attributes

The list of attribute groups used in the valuation studies, together with the data on the frequency of using these attributes in studies, is presented in Table 4.6. The most frequently used attributes are related to ecology and water quality improvements, followed by increased recreational suitability and improved aesthetics. Flood and erosion control are not routinely used in assessing the benefits of the river restoration studies.

Location, climate, and demographic characteristics

A brief summary of river location hydrologic and climate characteristics, such as catchment area, annual precipitation, and average annual temperature, is given in Table 4.7 for different world regions. In addition, we report some respondent-related information, such as annual average household income, average sample size, and average population density. The main conclusion is that the locations are different enough to ensure that the meta-analysis results are valid for a broad range of river restoration projects, although such variability will likely lead to a higher variance of modeling errors.

Table 4.6. Distribution of articles and estimates across valuation attributes

Attributes	Number of articles	Number of WTP estimates
Ecological improvement	22	63
Water quality improvement	21	54
Flow rate increase	10	23
Erosion control	7	11
Local economic impact	12	29
Flood risk reduction	3	7
Better aesthetics	9	30
Recreational suitability	15	39

Table 4.7. Summary of selected location, climatic and demographic variables

Parameters (average over sites)	Europe	Asia	America
Household income (annual, € 2013)	35,811	15,654	42,744
Sample size	446	472	419
Population density (p/km^2)	213	2212	427
Catchment area (km^2)	199,901	66,687	11,879
Annual precipitation (mm/yr)	696	1115	818
Average temperature (Celsius)	10.5	14.8	13.7

4.4 Econometric model and statistical estimation methods

The database consists of N studies, and each study i provides some varying number $j = 1,\ldots,J_i$ of estimates, or "effect-sizes" y_{ij} of the effect of interest, in this case mean WTP for river restoration. Each study also provides a standard error for this estimate σ_{ij}, which we assume is known (for example, reported in the original studies), and a number of explanatory variables $\mathbf{x}_i$ (covariates), which usually vary only across studies and provide information on studies, samples, characteristics of the environmental goods under valuation, and location characteristics. Most regressors are categorical variables and so specified as binary dummies.

In the description below we follow the taxonomy of meta-analysis models proposed by Nelson and Kennedy (2009). In particular, the models are classified with respect to sample heterogeneity, heteroskedasticity of effect-size variances, and non-independence of effect-size estimates. The first issue is addressed by restricting the data for meta-analysis in such a way that the included effect-sizes are uniform (e.g. mean WTP estimates only).

There are several ways to deal with the second issue of heteroskedasticity. It can be ignored, as is the case in the fixed-effect-size (FES) meta-analysis model, assumed completely random, as in the random-effect-size (RES) meta-analysis model, or fully explainable by explanatory variables, as in a FES meta-regression model. However, a better way to deal with the heteroskedasticity is to assume it is only partially explainable by covariates, which is conceptualized by the so-called random-effects meta-regression, or mixed-effect-size (MES) meta-analysis model, described in Sutton et al. (2000). Unlike the FES meta-regression model, it assumes that the variation present in the data can be explained by regressors only partially, either because of unobservables or because the effect-sizes are drawn from a distribution of population effects. This model, in which all individual effect-size estimates are considered independent, is defined as:

$$y_i = \mathbf{x}_i\beta + u_i + \epsilon_i, \quad u_i \sim N(0,\tau^2), \quad \epsilon_i \sim N(0,\sigma_i^2) \tag{4.1}$$

where $\mathbf{x}_i$ is a $1\times K$ vector of covariate values in study i, β is a $K\times 1$ vector of coefficients, ϵ_i is the measurement error term with variance σ_i^2 defined by the reported standard errors of y_i estimates, u_i is the error term accounting for the unexplainable heterogeneity of WTP estimates, and τ^2 is the between-study variance to be estimated from the data. Thus the variance of the composite error is $v_i^2 = \sigma_i^2 + \tau^2$, and can be used for estimation of the MES model by generalized least squares regression analysis.

As noted by Harbord and Higgins (2008), random-effects meta-regression may be considered either as an extension to fixed-effects meta-regression that allows for residual heterogeneity (i.e. between-study variance not explained by the covariates) or as an extension to random-effects meta-analysis that includes study-level covariates.

The third issue of correlated effect-size estimates often arises if primary studies in the database produce more than one estimate of the effect-size, as is often the case in economic valuation studies. In this case, the panel structure of the data (i.e. when y_{ij} is a j th-estimate drawn from study i) becomes important, and it is preferable to use panel data econometric models to account for the non-independence of the estimates:

$$y_{ij} = \mathbf{x}_{ij}\beta + u_i + \epsilon_{ij}, \quad u_i \sim N(0,\tau^2), \quad \epsilon_{ij} \sim N(0,\sigma_{ij}^2) \tag{4.2}$$

These models can be estimated by random-effects (RE), fixed-effects (FE), or between-effects (BE) estimators. The panel data models have several advantages in addition to accounting for the non-independence of effect-size estimates. The FE-estimator allows to avoid a bias due to a possible correlation between the heterogeneity term u_i and the regressors. Harbord and Higgins (2008) note that a MES model is closely related to the BE-model, which alleviates measurement error (by averaging data within each study) and is thus preferable in the presence of anchoring in CV studies (see also Nelson and Kennedy (2009):

$$\overline{y_i} = \overline{\mathbf{x}_\mathbf{i}}\beta + u_i + \overline{\epsilon_i}, \quad \overline{y_i} = \sum\nolimits_j y_{ij} / J_i, \quad \overline{x_i} = \sum\nolimits_j x_{ij} / J_i, \quad \overline{\epsilon_i} = \sum\nolimits_j \epsilon_{ij} / J_i \tag{4.3}$$

where the bar over the variables indicates averaging, and J_i is the number of effect-size estimates in each study. This model is usually estimated by weighted least squares (WLS), with the BE residual defined as $\tau^2 / J_i + \sigma_\epsilon^2$.

However, the ME-model is more useful if at least one variable is categorical with a set of discrete levels, as is almost always the case for meta-analysis. The model allows to include random effects (z_{ij}) other than those associated with the error term:

$$y_{ij} = \mathbf{x}_{ij}\beta + \mathbf{z}_{ij}u_i + \epsilon_{ij}, \quad u_i \sim N(0,\tau^2), \quad \epsilon_{ij} \sim N(0,\sigma_{ij}^2) \qquad (4.4)$$

This model is flexible enough to incorporate both fixed-effects parameters (x_{ij}) and random effects (z_{ij}), and this is why we selected the ME-model for our analysis. The model is estimated using the maximum likelihood method under the assumption that the random effects are uncorrelated.

4.5 Model Estimation

Univariate meta-analysis

The distribution of mean WTP estimates across the entire database is skewed, with the mean value being €69.9 per household per year and the median €43.1 per household per year (see Figure 4.6). Although there are some differences in WTP estimates averaged across world regions, e.g. €66.5 for Europe, €64.0 for Asia, and €76.9 for America, statistical tests does not indicate that there are significant differences between these values. The Kruskal-Wallis test for the equality of the mean WTP distribution across different regions reports a *p*-value of 0.60. At the same time, as Figure 4.7 shows, there is much more variation at individual country level, with mean WTP ranging from €11.3 for Korea to €118.0 for Scotland.

Comparing the mean WTP values across different elicitation methods, we find that the average WTP value derived from choice experiments (€95.5) is significantly higher than the average WTP value for contingent valuation studies (€52.3), with a *p*-value for the associated Kruskal-Wallis test statistic equal to 0.008. However, differences in average WTP values for the different CV elicitation formats are contrary to expectations not statistically significant.

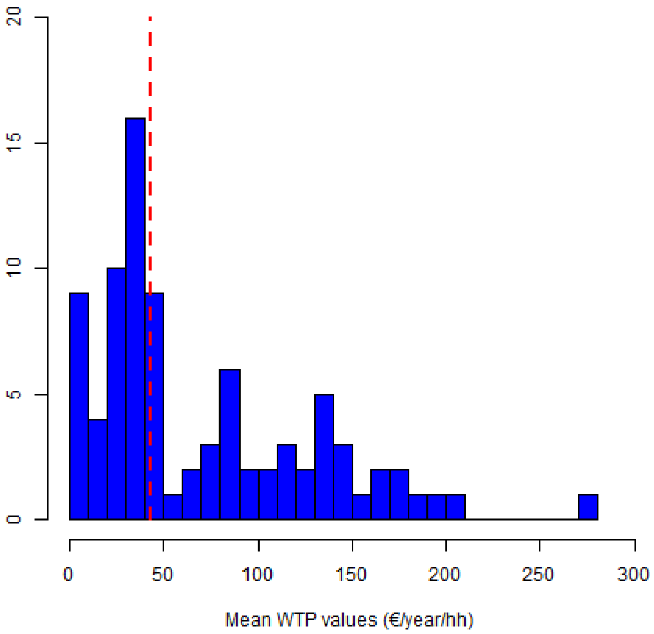


Figure 4.6. Histogram of mean WTP values for river restoration across all regions, in 2013 euro prices per household per year (red line indicates the median WTP estimate)

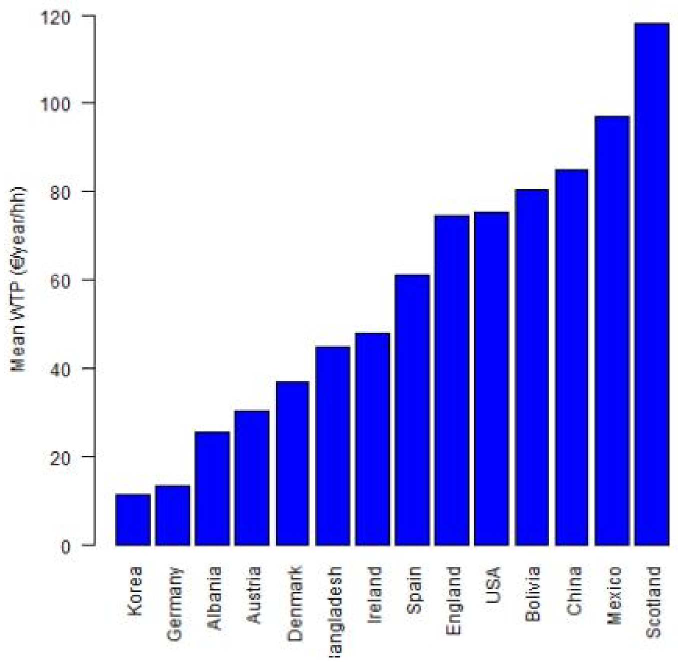


Figure 4.7. Ordered mean WTP distribution across countries, in 2013 euro prices per household per year

Multivariate meta-analysis

A mixed-effects multivariate regression panel model was estimated to test the influence of covariates simultaneously and address both between-study and estimate heterogeneity. For the multivariate meta-analysis we use 29 groups (studies) with 107 individual data entries (WTP estimates) in the database. In the process of model selection, several models are estimated that include the main characteristics of the river restoration project, the ecosystem services involved in the valuation scenarios, and the socio-demographic characteristics of the respondents. Categorical variables are coded as dummies, and the continuous variables, such as estimated WTP, average household income, population density, and fraction of the river length studied in a particular river restoration project, are transformed into their natural log form to improve the model fit, and allow for easy interpretation of the coefficient estimates.

The estimation results for the statistically best-fit model, which includes characteristics of the river and ecosystem services, site and population characteristics, as well as characteristics of the valuation method, are presented in the first column of Table 4.8. The overall fit of the model is very good, and the fixed effects explain 68 per cent of the observed variance. Compared to provisioning services such as drinking water and irrigation water supply (the baseline category in the estimated models), WTP for the regulating service flood control is significantly lower and WTP for the regulating services water quality control and erosion control significantly higher. All else being constant (ceteris paribus), mean WTP for the cultural services river recreation and landscape aesthetics (role of restored rivers in landscape valuation) is significantly higher compared to provisioning services.

Only in the reduced model do we find a significant positive effect for the fraction of the river that is being restored. However, once we include control for the ecosystem services, this effect becomes insignificant. EU respondents have a significantly lower WTP than respondents elsewhere in the world (US, Asia, Latin America). Also, WTP in more densely populated areas is, as expected due to higher overall demand, significantly higher. Higher income results as expected in a significantly higher mean WTP in the full model. Unfortunately no significant differences are found between users and nonusers.

With respect to the methodological study characteristics, discrete choice experiments generate significantly higher WTP values than CV studies, all else being constant. No significant differences exists between face-to-face (the baseline category) and web-based surveys. Mail surveys, however, generate significantly higher WTP values for river restoration than face-to-face interviews. When asked to pay on behalf of someone's entire household, this significantly reduces mean WTP compared to asking for someone's individual WTP (the baseline category). No significant effect of payment frequency can be detected, which is contrary to the temporal embedding effects observed for example by Stevens et al. (1997), Kim and Haab (2003), Spaninks and Hoevenagel (1995), and Brouwer et al. (2008). Also, a significant effect is found for payment vehicle: conform findings in Brouwer et al. (1999) for wetland ecosystem services, taxes reduce WTP significantly compared to other payment vehicles such as fees (e.g. entrance fee).

Table 4.8. Estimated meta-regression models

Variable	Full meta-model	Reduced transfer model (1)	Reduced transfer model (2)
Intercept	-0.798	1.092	0.358
	(2.139)	(2.018)	(2.937)
River and location characteristics			
Location (Europe=1)	-0.991**		
	(0.418)		
Restored river fraction (0-1)	-0.173	1.178*	0.771
	(0.506)	(0.606)	(0.796)
Population density (people/km^2)	0.309***	0.016	0.178
	(0.079)	(0.102)	(0.110)
Population characteristics			
River user (dummy)	0.245		
	(0.278)		
Average income (€/yr)	0.349*	0.196	0.085
	(0.199)	(0.196)	(0.278)
Valued ecosystem services			
Water quality control	1.602***		1.238***
	(0.247)		(0.268)
Flood protection	-2.978***		-3.585***
	(0.408)		(0.455)
Erosion protection	0.418*		0.352
	(0.238)		(0.261)
Recreational amenities	0.400**		0.287
	(0.188)		(0.201)
Landscape aesthetics	0.759***		0.716***
	(0.159)		(0.168)
Biodiversity	0.255		0.127
	(0.195)		(0.210)
Study characteristics			
Valuation method			
Choice experiment	0.589**		
	(0.299)		
Administration mode			
Web-based survey	0.042		
	(0.509)		
Mail survey	1.059***		
	(0.400)		
Payment characteristics			
Household (instead of individual)	-1.699**		
	(0.665)		
Payment frequency	-0.349		
(1 = less than annual)	(0.394)		
Payment vehicle			
Water bill	-0.358		
	(0.391)		
Tax	-1.411***		
	(0.451)		
Income tax	-3.465***		
	(0.904)		
Model summary statistics			
Log likelihood	-94.8	-168.5	-112.8
R^2 (fixed effect)	0.68	0.09	0.49
R^2 (overall)	0.89	0.38	0.95
AIC	233	349	249
Number of observations	107	107	107

Note: *p<0.1; **p<0.05; ***p<0.01

We also estimated the smallest possible reduced meta-regression models, for value function transfer purposes, for which the results are presented in the second and third columns of Table 4.8. If we only include easy measurable variables based on available secondary data sources like the fraction of the river that will be restored, population density and income, only the first variable is significant. This result is interesting: the higher the share of the river restored, the higher WTP (=sensitivity to scope). Although positive, the estimated coefficients for income become insignificant. Also the effect of population density disappears.

There is also a reduced form model that includes the ecosystem services (reduced model 2). Notably, this model shows much better fit compared to the reduced model 1. In this case the same ecosystem services are significant again except recreation and erosion protection. And only population density is marginally significant, as the fraction restored becomes insignificant, and income remains insignificant.

4.6 Conclusions

In this conclusions section, we report the transfer errors for the full best-fit model and the two reduced models, and compare these estimates with the transfer errors for the fixed-effect-size model, i.e. when we take the average WTP to be the best predictor for observed WTP estimates, and there is no need to include any control for other explanatory variables. This allows us to conclude how good the models are in terms of predictive power to assist in future benefit transfer exercises and support policy and decision-making.

The transfer errors are calculates as out-of-sample relative prediction errors, where one observation is omitted from the sample, the model is re-estimated, and a new predicted WTP value is calculated. The resampling is done by a jackknife procedure for each meta-analysis model. Table 4.9 reports the average results (mean, median, and standard deviation of transfer errors) that are based on the jackknifed samples, i.e. across all possible one-entry data omissions. The most notable result is that the full regression model reduces the prediction error by an order of magnitude compared to the simple average WTP model, and substantially reduces error variance of the predicted WTP values. The second reduced model that includes the variables for the ecosystem services also performs well compared both to the average WTP and first reduced models.

Hence, including control for fraction of the river that is restored, population density and income reduces the prediction error by almost a factor 3 compared to simply transferring mean WTP values. Adding in control for the ecosystem services further reduces the prediction error by almost a factor 4. The full model yields the lowest prediction error of, on average, 30 percent.

We also test for differences in sampling distributions of mean transfer errors for different meta-analysis models. Several two-sample tests, such as Wilcoxon and Kruskal-Wallis, deliver mostly comparable results. First, the difference between average transfer errors for the simple FES model and for the full ME model is highly significant (p-value is less than 0.01), indicating that the latter significantly outperforms the former. Similarly, the differences in mean transfer errors for the FES model and for any of the reduced models are significant at 0.01-level. However, the evidence for the differences between the full and reduced models is somewhat mixed, as different tests lead to conflicting conclusions about the significance of differences in mean transfer errors in this case.

Table 4.9. Transfer errors for different models

	Mean WTP model	Best-Fit full model	Reduced model 1	Reduced model 2
mean	10.85	0.31	4.02	1.07
median	0.53	-0.09	-0.16	-0.16
std. dev.	53.88	1.22	20.90	4.92

5 Key methodological issues in CBA and practical case study illustrations

5.1 Introduction

Current scientific understanding of river rehabilitation is generally poor (Vaughan et al. 2009), many uncertainties still arise and there is still limited understanding of how river systems and catchments respond to rehabilitation (Szaro et al. 1998; Downs & Kondolf 2002; Gillilan et al. 2005; Jansson et al. 2005). While there is a steady increase of restoration projects each year, the absence of adequate monitoring and evaluation is most frequently a result of a lack of resources than unwillingness to do so and this constrains the ability to assess the effectiveness of rehabilitation techniques (Eden & Tunstall 2006; FAO 2008). In addition, the benefits generated by rivers are difficult to quantify and evaluate, but this is fundamental to undertake a CBA of the most appropriate measures for achieving the outcomes as defined in the WFD. An economic appraisal of the whole project and not only the cost of restoration measures is vital for resourceful river rehabilitation projects. The review of concepts to measure the success of river restoration found that despite large economic investments in what has been called the “restoration economy”, many practitioners do not follow a systematic approach for planning restoration projects. As a result, many restoration efforts fail or fall short of their objectives, if objectives have been explicitly formulated at all. Furthermore, the key problem to our paucity is poor project design and implementation, consequential to the outcome of restoration being intangible and difficult to quantify.

Monitoring and evaluation is a necessary process that should be included in all river project planning frameworks because it determines the effectiveness of rehabilitation actions in support of the WFD (WFD (2000/60/EC)) (Wolter 2010). There are various guidance manuals on how to design restoration projects and implement monitoring programmes (RRC 2011; Cowx et al. 2013; Roni & Beechie 2013), but the uptake of these methods is slow and without such analysis it is difficult to assess to what extent the restoration is successful (Possingham 2012). If social and biological benefits are not monitored then it is difficult to identify cost-effective restoration, carry out a comprehensive CBA and identify the level of project success.

Moreover, the aims of restoration activities in Europe are influenced by a plethora of EU Directives and national government policies that may have conflicting targets. The potential for restoring river ecosystems to achieve win-win situations for biodiversity and ecosystem services and to consider a much wider framework of environmental policy and practice is increasingly advocated as a rational approach for river managers. The planning process for these synergies has specific emphasis on the future management cycles to compile strategies for Programmes of Measures (PoM) to support River Basin Management Plans (RBMPs) and the tuning of the WFD with other directives such as the Habitats Directive (HD (92/43/EEC)), CBD (Convention of Biological Diversity) and Ground Water Directive GWD (2006/118/EC). Recent developments have resulted in potentially conflicting directives to the WFD, such as the Floods Directive (EU FD), the Renewable Energy Directive (RED) and the Sustainable Transport Directive (STD), all of which are necessary to support river management from a socio-economic perspective. Consequently, river restoration tends to encounter obstacles as a result of societal demands, particularly through a select number of ecosystem services, such as provisioning and regulating services like flood protection, hydropower, navigation and agriculture (discussed in D5.3). For example, there is growing conflict between land drainage and flood prevention works, as well as hydropower development, and the environmental lobby, who argue against rehabilitation because of increased flood risk. Nonetheless, the WFD indicates that all rivers must be returned to a good ecological status or achieve their best ecological potential by the year 2015 and if not then, by 2027. Therefore, it is imperative that synergies between measures are explored and the measures that deliver the greatest environmental outcomes, both directly and indirectly, are prioritised. The planning procedure should therefore include a CBA of the proposed project options that will maximise the benefits involved.

Currently, there exist only a very limited number of guidelines to judge and plan river rehabilitation using an integrated approach based on ecological, physical, sociological and economic considerations. Project planning that considers such an integrated approach, and allows for the identification of project success from an ecological, social and economic perspective is crucial in advancing river restoration practices. Details of the importance of a '*project planning framework*' is discussed in D5.1 (Cowx et al. 2013). The planning stage should identify the purpose and need for restoration through pre-monitoring where remedial action should focus on the underlying cause(s). More specifically pre-monitoring will evaluate watershed processes, current river health and ecological status to further: (1) identify how habitats have changed and altered biota; (2) identify the causes of habitat changes; (3) identify rehabilitation actions needed to address those causes; and (4) acknowledge social, economic, land use constraints and synergies (Beechie et al. 2008, 2009). This will enable suitable goals (define project intent) and objectives (specific and measurable outcomes) to be established for restoring the system to an acceptable state, ultimately leading to a self-sustaining river ecosystem (Cowx 1994; Kondolf et al. 2006; England et al. 2007).

Social cost-benefit analysis is, or can be, an important tool in project planning. This chapter discusses therefore some of the key methodological issues in applying social cost-benefit analysis to river restoration projects and presents a number of practical examples from actual CBA's of river restoration projects in Europe. These key methodological issues are directly related to the steps in a social CBA as outlined in the Introduction to this report, and focus in particular on:

(i) the scope of the analysis in the first step (Section 5.2),
(ii) the definition of the baseline and policy scenarios in step 2 and 3 (Section 5.3)
(iii) the identification of the project effects (Section 5.4), in particular indirect effects (Section 5.8) and the link between biophysical impacts and ecosystem services (Section 5.5) in step 5,
(iv) the monetary valuation of these ecosystem services based on non-market valuation techniques (Section 5.7) in step 6, and
(v) the comparison of costs and benefits to assess the potential disproportionateness of the costs (Section 5.9) in step 9 and 10.

5.2 Financial *versus* economic analysis

A financial analysis of a river restoration project measures all expected cash flows to and from the project 'developer', often a governmental authority, over the time horizon of the project and calculates the private rate of return. A financial analysis, for example as part of a *business case*, can be very useful to attract the financial means to carry out the project, to monitor progress, and to evaluate the key outcomes (Sayer Vincent 2009). A financial analysis accompanying a business case can be an important input into a Social Cost Benefit Analysis (SCBA), but the two types of analysis are fundamentally different.

In the first place, the financial analysis measures the cost and benefits for one party only (the developer of the project), while the SCBA should measure the costs and benefits to all parties involved. For a river restoration project, these parties benefitting from the project could include active users of the resource, people who value gains in biodiversity (users and non-users), or home-owners in the vicinity of the project. The costs usually accrue to tax payers and possibly to the owners of polluting activities that are being banned or regulated.

Related to the different perspective is that also the basis of valuation of goods and services and factors of production may differ between financial and economic CBA. In a financial CBA the basis of valuation is the market price for the developer, including indirect taxes and subsidies. In an economic CBA, the formal basis of valuation is the opportunity cost of the resources, that may or may not differ from the market price. A clear example is a subsidy on the price of some good or resource that is used in a river restoration project. In a financial CBA, a subsidy makes the good or resource cheaper so that the project result, from the perspective of the developer, is enhanced. In an economic CBA, however, the benefit to the project developer is offset by an additional cost to the government budget or the tax payer that must also be accounted for. In an economic CBA, therefore, a subsidy should either be accounted for on the cost and the benefit side, or be netted out of all financial flows. There can be differences between national guidelines on the exact definition of the prices that should be used in an economic CBA, e.g. with respect to including or excluding Value Added Tax (VAT). There can also be differences between national and EU guidelines. For river restoration projects, the appropriate national of EU guidelines should be used to determine what prices should be used. One particular ‘price’, is the discount rate that is used to calculate the ‘present value’ of future costs and benefits. In a financial CBA, the developer is free to choose his or her own discount rate, while in an economic CBA the correct discount rate is based on social preferences and usually prescribed by the government.

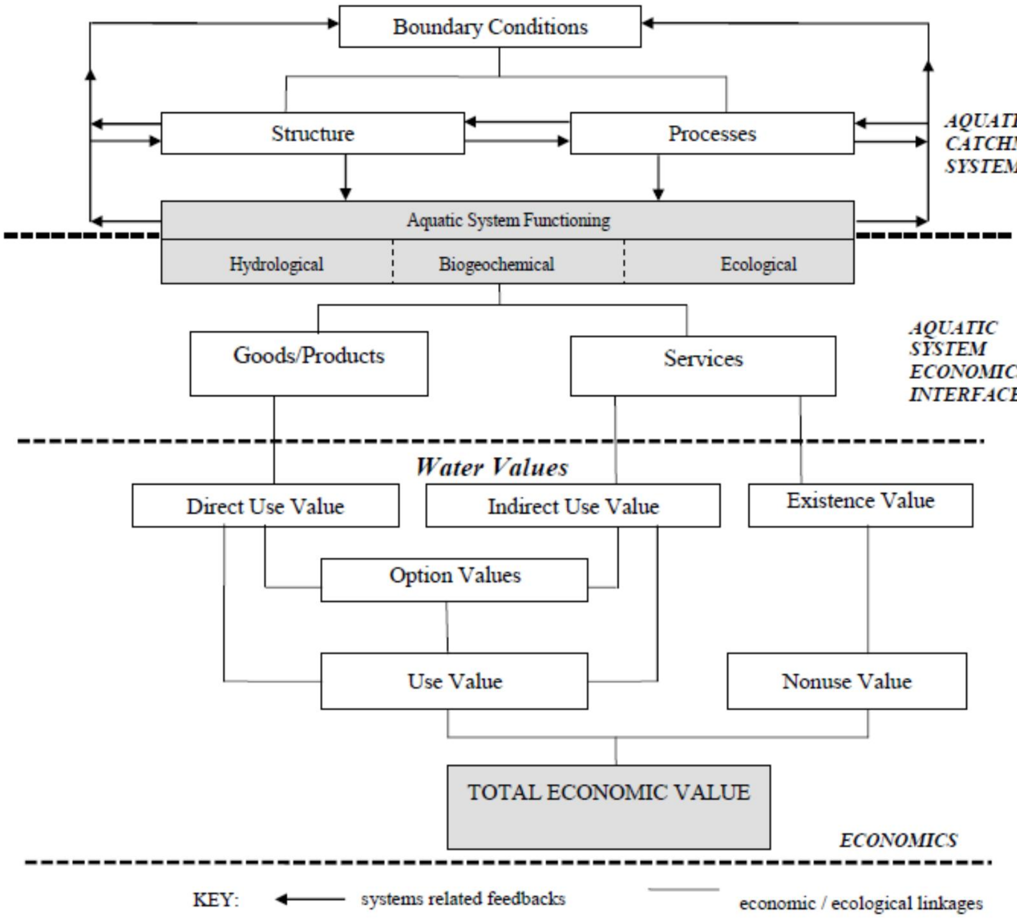
In the second place, a CBA tries to account for all flows of goods and services, whether they have a market price and can hence be monetized or not. While a financial analysis only takes into account “cash” flows, CBA takes account of cash flows and the flows of ecosystem services that may not be traded in markets and therefore do not have market prices.

The Head Weir Removal Project on the River Taw, North Devon, involved replacing an Alaska Denil pass with a modern Larinier pass on Head Weir. The main ecological benefits are increased access of salmon to a significant area of spawning habitat and access of lampreys for the first time ever; increasing the resilience of a very rare population of fresh water pearl mussels; improving water quality; and diminishing the risk of invasion of non-native species. The CBA of the project identified as the main beneficiaries local anglers who use the river for recreational angling and the local population (defined as those people living within 60 miles of the river) who will derive (non-use) benefits from knowing the biodiversity of the river will be improved. The benefits to these groups of beneficiaries involve no “cash” flows but are derived from the estimated willingness to pay of anglers and households for the respective ecosystem services.

In the absence of markets for many ecosystems associated with rivers it is necessary to have knowledge and information of the marginal value or benefits of the river resource in its alternative uses. The economic definition of value is cast in terms of economic behaviour in the context of supply and demand. It is the maximum amount of goods or service – or money income that an individual is willing to forego (willingness to pay or WTP) in order to obtain some outcome that increases his or her welfare. If the outcome reduces welfare then this utility loss is measured by the minimum amount of money that the individual would require in compensation (willingness to accept or WTA) in order to suffer the changes. These WTP/WTA amounts are demonstrated or implied by the choices people make (or say that they intent to make), and thus reflect individuals' preferences for the change in question (Brouwer et al., 2009).

Aggregated across those who benefit from a good or service and hence who will be affected by any change in their provision level, the aggregate WTP or WTA amount provides an indicator of a project's Total Economic Value (TEV). Economists have introduced a taxonomy of this TEV which captures the variety of values emanating from the different uses of environmental resources, including rivers. The aggregate WTP measure of the impact on social welfare does not consider inequalities in the distribution of gains and losses among individuals. However, WTP is constrained by individuals' ability to pay.

Figure 5.1 presents a general framework for water resource valuation from Turner et al. (2004) that makes use of the TEV concept. As can be seen in the bottom of the figure, TEV is comprised of use and nonuse values. Use values can be direct or indirect, and they can be actually enjoyed or they are valued because of the *option* to be enjoyed in the future. People who do not use a particular resource and do not have the intention of ever using it, may still value it because of its mere existence. This is a non-use value and in Figure 5.1 it is directly linked to the Existence Value. Figure 5.1 also shows how economic valuation is linked to the an ecosystem functional approach.



Source: Turner et al. (2004).

Figure 5.1 Framework for water resource valuation

5.3 Definition of the baseline and project scenarios

Project identification provides an understanding of the current status of the ecosystem functioning and ecosystem services in the management zone to establish the baseline situation against which to develop a restoration project. Key to this evaluation is the assessment of the interrelationships between human activities and environmental factors that drive the ecosystem functioning and provision of services. The basic information required includes, but is not exclusive to:

- Background geography and landscape topography, political domains, climate and general infrastructural development;
- Habitat modification and geomorphological alterations;
- Hydrology, including modifications to flow regulation, abstraction and other water uses;
- Flood defence;
- Fisheries, recreation and conservation;
- Water quality;
- Land use/navigation and mineral extraction;
- Urban, agricultural and industrial development.

The importance of establishing an appropriate baseline is illustrated in the next two examples that are taken from the literature (e.g. Brouwer and van Ek, 2004; Brouwer and Kind, 2005). After serious threats of flooding in the Netherlands in 1993 and 1995, when more than 250,000 people had to be evacuated from areas along the rivers Rhine and Meuse, the Dutch government started to investigate alternative options to maintain existing flood protection and safety levels. Broadly, there are two main types of response options: traditional technical engineering approaches (dike strengthening) and an alternative approach of making use of the natural dynamics and resilience of water systems, that would involve land use change and floodplain restoration. A study on the cost and benefits of alternative flood control policies by way of land use change and floodplain restoration was carried out in the late 1990s (Brouwer and van Ek, 2004). In this study, the land use change and floodplain restoration option was compared to a 'do-nothing' baseline, implying that the current flood protection and safety levels would not be maintained in the absence of the project. An important benefit of the project was therefore the expected 'avoided damage' of additional flooding. However, because of the fact that flood protection and safety levels in the Netherlands have a legal status – they are fixed by law – the question is whether the 'do nothing' scenario is the appropriate baseline. In fact, one could argue that 'doing nothing' is simply not allowed by law and therefore the baseline should be the traditional option to maintain flood protection and safety levels, i.e. the dike strengthening option. In this case, the land use change and floodplain restoration option would still have benefits in terms of ecological and recreational benefits, but there would be no or significantly less 'avoided flooding damage' benefits. Figure 5.2 shows graphically the effect that a different baseline has on the amount of benefits in this case. In the 'do –nothing' scenario, the risk of flooding – expressed in expected damage costs – is expected to increase. In the 'dike strengthening' scenario, the risk of flooding is substantially reduced. In the 'floodplain restoration scenario' the risk of flooding is also reduced and there are additional ecological and recreational benefits. Figure 5.2 shows that the magnitude of the benefits of the assessed option critically depends on the baseline scenario.

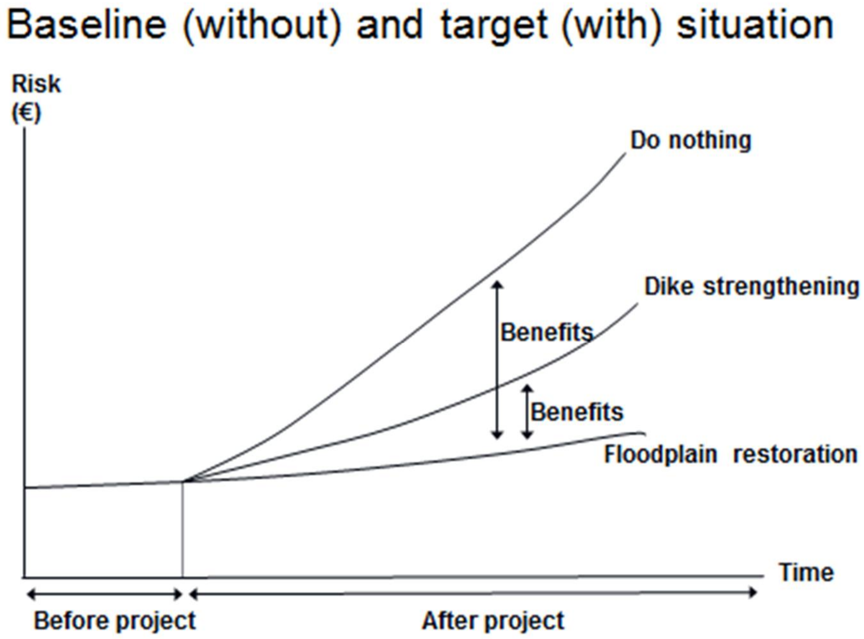


Figure 5.2 The importance of the appropriate baseline to assess the benefits of policy options

Another example from the Netherlands shows the importance of an appropriate baseline to assess the feasibility of policy options. Also in response to the flood threads of 1993 and 1995, a large study into flood control measures for the River Meuse was commissioned by the Dutch government. To assess the feasibility of spatial interventions (by-passes, overflow areas, etc.) a detailed study was made related to the expected (exogenous) socio-economic development in the area until 2050. In particular, this 'baseline' included the geographical assessment of residential development plans of municipalities, planned industrial sites, agricultural, recreational, and nature protection expansion plans. Maps with these exogenous developments were developed together with the various stakeholders involved (agriculture, industry, municipalities etc.) and compared with similar maps displaying the different spatial river restoration options to assess the feasibility of the options and possible bottlenecks and trade-offs (Ministry of Transport, Public Works and Water Management, 2003). Figure 5.3 shows the maps of the river restoration options (left) and the identified spatial developments under baseline conditions (right), i.e. independent of the planned policy interventions based on demographic and socio-economic trends and plans.

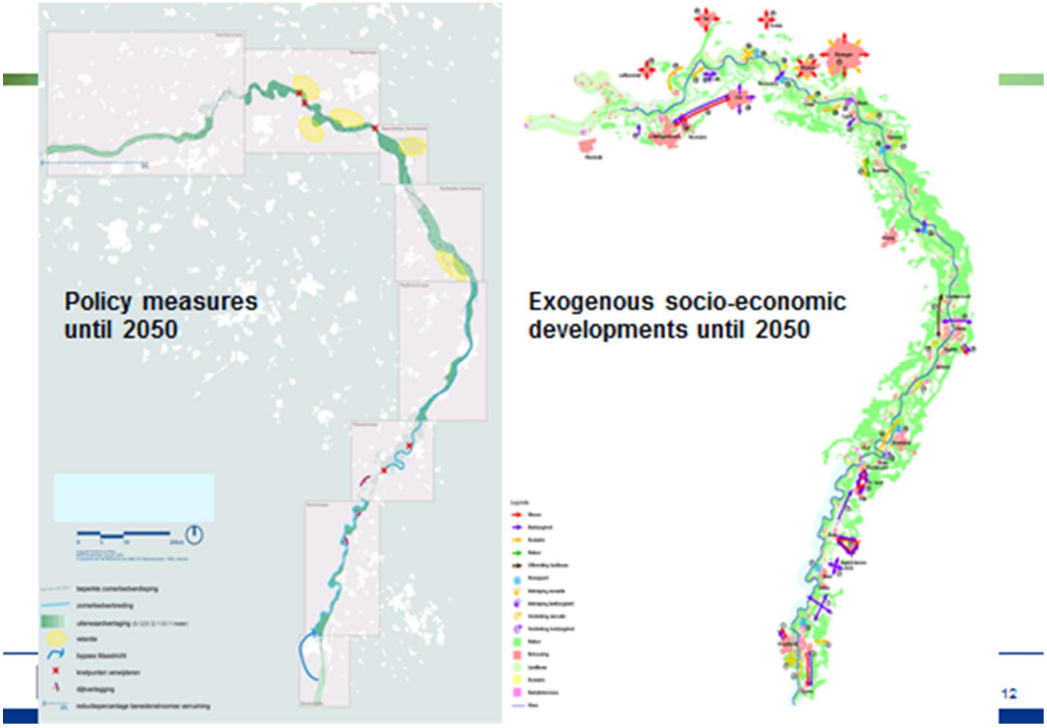


Figure 5.3 The importance of the appropriate baseline to assess the feasibility of policy options (source Ministry of Transport, Public Works and Water Management, 2003)

> In the Butter Mill Fish Easement project a Pan weir has been removed on the River Wandle in London. Additional trout habitat has been opened up and the aesthetic attributes of the river have been significantly enhanced. For the CBA of this project, as for all projects that were evaluated for the UK's River Improvement Fund, the assumption has been made that without the project the environmental quality of the river would remain constant over the evaluation period. Implicitly, it was also assumed that both the number of beneficiaries and their preferences would remain unchanged over the evaluation period. While these assumptions might have been appropriate for the projects under consideration, it would be recommendable for CBA analysts to reflect on these 'baseline' assumptions.

5.4 Identification of project outcomes

In order to assess project outcomes, restoration objectives should be clearly defined adopting a river basin-wide approach, and related to high priority WFD and national policy objectives. Setting benchmarks and end points that are linked to clearly defined project goals is considered the best approach to help determine the measure of success, especially when goals are linked to objective success criteria to guide the process and the likelihood of achieving the end result (Bernhard et al. 2007). Benchmarks and endpoints place a level of quality to rehabilitation that can be used as a standard when comparing other things against which to measure performance. They should be reviewed against reference conditions, to determine appropriate targets for restoration, rehabilitation and mitigation activities. However, river restoration, rehabilitation and mitigation require several areas of knowledge such as ecology, hydrology and engineering (Doyle et al.

1999) and goals relating to composition, structure, function and other ecological parameters, thus it is complex and considered difficult to define which measures should be used to quantify the success (Hobbs & Harris 2001). The meaning of 'success' will change depending on the type of water body, type of project, the condition of the river health and the ecosystem services it supplies. For example, areas of HMWBs need only reach good ecological potential and therefore will have different endpoints and measures of success. It may be more achievable to reach a level of success when the goal is to restore a certain level of function/species rather than to attempt complete restoration (Lockwood & Pimm 1999) and therefore realistic goals are essential for progress towards success (Hobbs & Harris, 2001; Hobbs 2007). The concept of increasing habitat heterogeneity to increase biodiversity through rehabilitation has been a long-standing approach (Jungwirth et al. 1995; Kondolf & Micheli 1995; Montgomery 1997; Palmer et al. 1997; Kemp et al. 1999), but this may not always be the most cost-effective approach. Introducing the design of benchmarking and endpoints in the planning stages will strengthen rehabilitation practices as it steers away from ambiguous proposals towards a more definite ideal of the required ecosystem in a specific segment of river. One other problem that needs to be overcome is to ensure compliance with endpoints and benchmarking related to other EU Directives, to ensure healthy aquatic ecosystems while at the same time ensuring a balance between water and nature protection and the sustainable use of natural resources is critical.

Benchmarking as a tool should be feasible, practical and measureable to help guide future decision support tools. Benchmarking uses representative sites otherwise known as 'reference sites' on a river that have the required ecological status and are relatively undisturbed; this is then used as a target for restoring other degraded sections of river within the same river or catchment. This approach therefore uses appropriate undisturbed sites of the same river type (Rheinhardt et al. 1999), rather than attempt to create conditions unrelated to the original ones at the site of interest and is consequently more likely to result in long-term success (Choi 2004; Palmer et al. 2004; Suding et al. 2004; Woolsey et al. 2007). The use of reference reaches to help restoration objectives is common in North America (Wheaton et al. 2004), but less common in Europe and other parts of the world where un-impacted reference reaches are rare (Statzner et al. 2005; Comiti et al. 2009; Skidmore et al. 2013 *in* Roni & Beechie 2013).

It is imperative that endpoints accompany benchmarking in the planning process to guarantee the prospect of measuring success because endpoints are feasible targets for river rehabilitation. It is important to note that endpoints are different to benchmarks, this is because other demands on the river systems also have to be met and references can only function as a source of inspiration on which the development towards the endpoints is based (Buijse et al. 2005). Part of the problem is that the ecological status or potential of a water body is used as the target status of the restoration measure and the biological quality elements are not necessarily sensitive enough to detect the change (Bernhardt et al. 2005; Palmer et al. 2010; Violin et al. 2011; Stranko et al. 2012, but see also Lorenz et al. 2012; Haase et al. 2013). Good ecological status or potential is intended to describe the extent to which ecological quality deviates from what would be expected under near natural conditions and should not necessarily be the goal of restoration; it fundamentally needs better formulated targets or end points. Given that benchmark standards cannot always be achieved, especially on urban rivers, endpoints will therefore assist in moving restoration effort towards benchmark standards through application of the SMART approach (WP5.1 Cowx et al. 2013) to decide what is achievable and what is feasible. There is a need to distinguish endpoints for:

- Individual measures;
- Combination of measures;
- Catchment water bodies;
- River basin districts.

It is important to recognize what is the minimum acceptable achievement level of restoration and what is the desirable level to have as a target end point that is still below the benchmark level, yet still aims for WFD status targets. Subsequently, what can be compromised for this desired level, will it be cost, ecosystem services or ecological aspects? Albeit, applying benchmarking to increase the accuracy and success of restoration appears in theory to be an uncomplicated method, in fact it increases the level of intricacy that rehabilitation needs to apply. This is because natural instream habitats consist of complex multidimensional arrays of morphological conditions (substrate, woody debris, hydraulic patterns) along with the complex life structures and habitat guilds of the biota (Statzner et al. 1988; Strange 1999) and the environmental conditions (velocity, depth, temperature) and resources (food, space) on which they depend, all of which need to be incorporated in to river rehabilitation. Using this example the process of benchmarking can be broken down into a number of steps:

- “Reference condition”: Deriving reference criteria – need to establish reference conditions of specific river types or river styles as defined by WP2. This may not be the pristine state but should describe the state or value of a defined ecological attribute if the system had not been disturbed by the specific pressure of pressures. It may well be defined by nearby undisturbed (by said pressure[s]) reaches of rivers that is achieving GES or GEP, i.e. an ecosystem with ecological integrity commensurate with that meeting societal aspirations.
- “Expectation”: Transfer reference conditions to end points for target systems – different for each river style including temporal and spatial dimensions. This will require comparison of status against objectives for restoration that are appropriate to accommodate variability in river style/types (WP2). Establishing endpoints identifies characteristics of concern that reflect the overall restoration goal.
- “Baseline condition”: Undertake deficit analysis (to identify what hydromorphological limitations and processes are constraining the recovery of the biota) and explore the potential for restoration to establish ‘endpoint’ target conditions.
- Once the end points have been established these restoration targets need integration into wider catchment-based activities to deliver win-win scenarios (e.g. flood mitigation, hydropower, agriculture, navigation) and take due account of the cost and benefits, specifically in relation to ecosystem services delivery, to ascertain the most effective measures to meet specific objective.

The timeframe over which monitoring programs are implemented should capture the natural range of behaviour of the river to show the timeframe over which geomorphological adjustments occur (Brierley et al. 2010). However, it is difficult to foresee the recovery time-scale for any rehabilitation project, especially those based around geomorphological modifications. When physical structures are installed in river channels to improve fish habitat, the adjustment process that occurs over time can sometimes be more harmful than do good (Rosgen 1994). Ecological recovery time from this type of habitat modification depends on hydromorphological characteristics of the river (Brookes 1996; Sear et al. 1998) and how this further affects ecological processes within the river; for this reason long-term monitoring is needed to enhance understanding (England et al. 2007). Recognizing when monitoring should take place is vital to increase the accuracy and understanding of the success level of each rehabilitation project. Both pre and post monitoring is essential within a river rehabilitation project planning framework. Pre-monitoring includes the collection of baseline data to assess the status of river health and fisheries health, and assist in the identification of river rehabilitation objectives (Kondolf & Downs 1996). Baseline data (or pre-monitoring data) can be used within river rehabilitation assessment to compare the status of habitat and fisheries of the river between pre and post monitoring of the rehabilitation works. Evaluating multiple control sites across a spatial scale will allow the level of success of rehabilitation projects to be measured by taking in to account patch dynamics (Clewell & Rieger 1997) to give a comprehensive review of the biota local to that river. Post-monitoring is an essential phase that is needed to assess the success of rehabilitation works, and long-term, post-monitoring will provide a more valuable data source for evaluation purposes; however, it is not always easy to know the length of monitoring needed but it should cover at least 2 generations of the longest living species (Kondolf & Micheli 1995). This should always be considered when costing a project.

Overall, river restoration schemes may cause negative economic impacts on certain economic activities by changes in water management and land use and thus, impede sectors such as navigation or agriculture. In order to support river basin planning and decisions, river managers need to assess these impacts with appropriate and transparent methods, and weigh them against the benefits predicted. This will require detailed consideration of regulations and socio-economic constraints at local, regional and national levels. Improving river ecosystem quality while maintaining or enlarging the wider socio-economic benefits by assessing strategies to support synergies between typical conflicting sectors is a much needed approach. A well designed restoration project will reduce the uncertainty of management actions through the implementation of policies and application of a logical path that links rehabilitation goals, watershed assessment, identification of rehabilitation needs, selection and prioritization actions, design of projects, and development of a monitoring program. Evaluating how successful restoration measures have been, as well as determining reasons for success or failure seem essential if restoration measures are to be carried out in an efficient and cost effective manner, especially in the European context with respect to meeting obligations under the WFD.

5.5 From ecological impacts to ecosystem services

Rivers that are not affected by hydromorphological (HYMO) pressures and that have free flowing conditions are able to support ecosystem services such as fish yield, floodplain agriculture, wildlife and biodiversity. Traditional water management, in contrast, has promoted the provision of ecosystem services such as hydropower and irrigation that depend on the construction of extensive infrastructure (Auerbach et al. 2014). This historical management has degraded fluvial ecosystems and decreased the flow of the above-mentioned ecosystem services of previously free-flowing rivers. It is obvious that ecosystem service assessments can and should account for ecosystem service benefits in the absence of water infrastructure to inform balanced water policy and watershed management (Auerbach et al. 2014).

Traditional water management has been the result of societal driving forces dominated by technological and economic motives. Nowadays, these driving forces also include environmental motives, leading to proposals for restoration measures. Figure 5.4 shows a scheme of the natural and socio-economic dimensions that underlie the complex interactions between driving forces, pressures and ecosystem services. Restoration measures are implemented in order to mitigate the effects of pressures. Although Good Ecological Status (GES) is defined by biological elements, restoration is not done by directly manipulating these biological elements. We do not stock fishes and macroinvertebrates; neither do we grow macrophytes in order to achieve GES. On the contrary, restoration is done by improving fluvial habitats by means of changing HYMO conditions.

When fluvial habitat is degraded, there are often several factors that limit the presence of reference biotic communities. These limiting factors may be pollution, insufficient flowing water, poor substrate, spatially homogeneous habitat, or lack of temporal variability. However, these factors do not have the same impacts on the biological elements. Each one has a different threshold and therefore, limiting factors are hierarchized in a way that, once a more restrictive factor is mitigated, another factor becomes the limiting one. In D1.3 Wolter et al. (2013) present this hierarchy of habitat bottlenecks. They show that when water quality (especially anoxic conditions) is the most restricting factor, that often hides HYMO factors.

After water quality the next limiting factors are of HYMO character: a) water quantity (water abstraction and flow regulation) b) habitat complexity (fragmentation, channelization,..); and c) key habitats (gravel pits,..). This bottleneck hierarchy is one of the reasons why many restoration projects are not effective in increasing rivers' ecological status: measures are designed to mitigate the main controlling factors, but do not address the underlying ones.

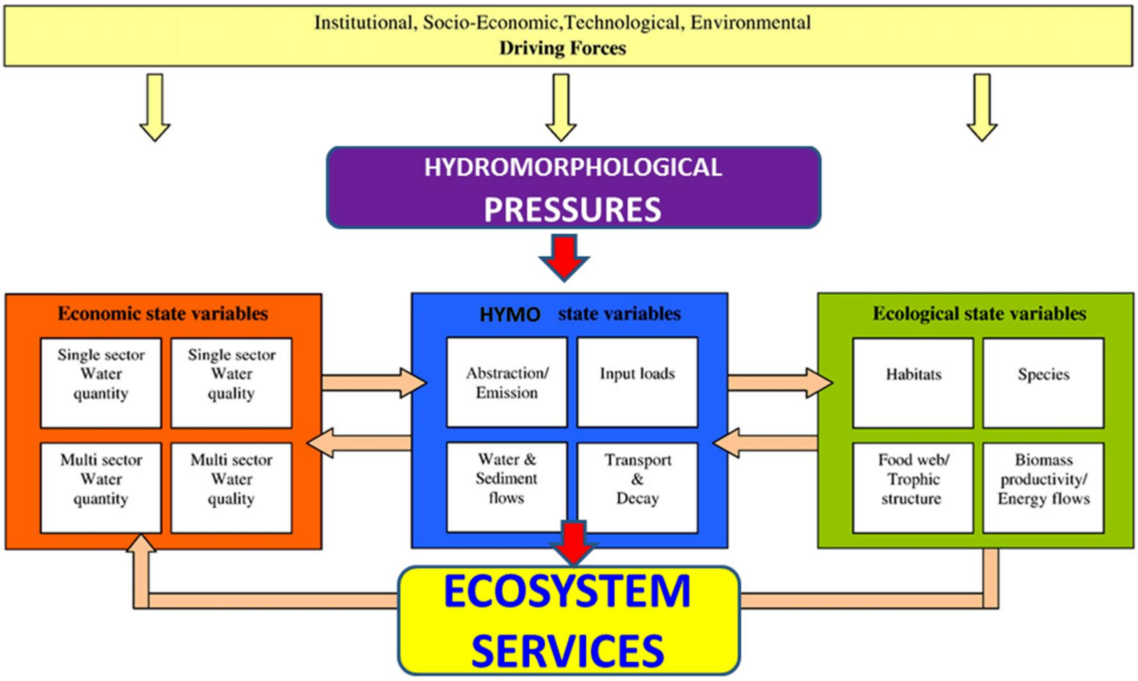


Figure 5.4. Scheme of the disciplinary dimensions underlying the complex interaction between driving forces, pressures and ecosystem services (modified from Brouwer & Hofkes, 2008)

Apart from these thresholds, the link between habitat improvement and the response of the biological communities is not well stablished. Other causes like biotic interactions, growth delay, behavioral responses, or natural variability may inhibit improvements of the ecological status. When restoration measures to mitigate certain impacts are applied, and the pressures that caused that impact have not been eliminated, there will be a compensation effect that may not be sufficient to pass the respective threshold.

In natural and free flowing rivers, changes in their normal functioning are caused by natural disturbances, such as floods, droughts, or geological events (Figure 5.5). The effects of these events are compensated by the alteration of HYMO processes that produce changes in their habitats and consequently in the biotas. However, the changes are not permanent as the resilience capacity of the ecosystem will produce a reversal tendency. Thus, the system has an oscillatory trajectory that includes all these changes which are the natural variability of the ecosystem and represents an important item of its natural biodiversity. This natural river functioning provides ecosystem services that may be considered as a reference.

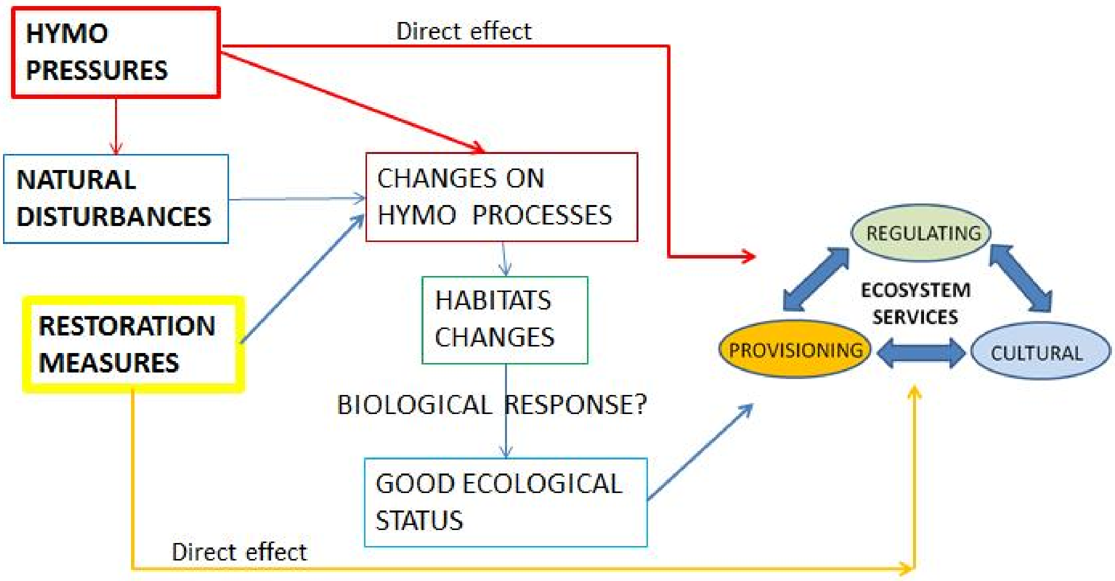


Figure 5.5 Diagram showing mechanisms how fluvial Ecosystem Services may be affected by Natural disturbances, HYMO pressures and Restoration Measures. Direct effects are simple to predict, while precise predictions of overall interactions affecting HYMO processes, habitat changes and the response of the biological elements requires much more science than available.

However, at present rivers are subject to anthropic pressures that degrade the status of the fluvial ecosystem and directly affect its ecosystem services. Some of these pressures are HYMO pressures as they alter the HYMO processes that regulate the river functioning. HYMO pressures impact biological communities as they change their habitats into others at which they are not adapted to. As a consequence, biological response is based on biodiversity reduction or changes in composition that promote invasive and alien species. Also, HYMO pressures may directly affect some ecosystem services (Figure 5).

On the contrary, restoration or mitigation measures are designed to improve habitats, either directly through structural measures, or through the recovery of lost HYMO processes. But also, sometimes restoration measures are targeted to recover some ecosystem services.

Understanding how a river with some natural variability works, simultaneously subjected to different pressures and programs of measures involves a serious difficulty. However, to be efficient and effective, the management of water bodies needs to predict the overall effects of the pressures and measures.

Table 5.1.- Main HYMO pressure types classified according to WFD Hydromorphological elements

Water Abstraction	Groundwater abstraction
	Surface water abstraction
	Inter-basin flow transfers (donor basin)
	By-pass: Returned flow downstream
	By-pass: Water abstracted is consumed
	Flood deviation
Flow regulation	Inter-basin flow transfers
	Hydrological regime modification (flow timing)
	Hydropeaking
	Reservoir flushing
	Sediment discharge
River fragmentation	Artificial barriers upstream from the site
	Artificial barriers downstream from the site
	Collinear connected reservoir
Morphological alterations	Impoundment
	Large Dams and Reservoirs
	Channelization: Cross section alteration
	Channelization: Channel realignment
	Alteration of riparian vegetation
	Alteration of instream habitat
	Embankments, levees or dikes
	Sand and gravel extraction
	Floodplain Soil Sealing and Compaction
physico-chemical preassures	Thermal changes
	Eutrophication (Nutrient enrichment)
	Organic discharge

Although multiple pressures affect rivers simultaneously and at the same time, for practical reasons we have distinguished single river pressures and their most direct impacts on ecosystems.

The pressures have been grouped into the following classes, bearing in mind the WFD HYMO elements and processes affected:

1. Hydrological regime
    1.1 Water abstractions
    1.2 Flow regulation

2. River fragmentation
3. Morphological alterations
4. Other elements and processes affected (physico-chemical)

Table 5.1 shows a detailed composition of each pressure type. In order to mitigate and achieve a good ecological status (or good potential in the case of HMWB) RBMP design a program of restoration measures. These measures have been classified in different types (following WP3):

- Water flow quantity improvement
- Sediment flow quantity improvement
- Flow dynamics (water and sediment) improvement
- Longitudinal connectivity/continuity improvement
- River bed depth and width variation improvement
- In-channel structure and substrate improvement
- Riparian zones improvement
- Floodplains/ off-channel/ lateral connectivity habitats improvement

Table 5.2.- List of main Restoration Measures types classified according to WFD Hydromorphological elements

Water flow quantity improvement	Reduce surface water abstraction without return Reduce surface water abstraction with return Improve water retention (catchment, basin) Reduce groundwater extraction Improve/Create Water storage Increase minimum flows Water diversion and transfer Recycle used water Reduce water consumption
Sediment flow quantity improvement	Add/feed sediment Reduce undesired sediment input Prevent sediment accumulation in reservoirs Reduce erosion Improve continuity of sediment transport Manage dams for sediment flow Trap sediments
Flow dynamics (water and sediment) improvement	Ensure minimum flows Establish environmental flows Modify hydropeaking Increase flood frequency and duration in riparian zones or floodplains Reduce anthropogenic flow peaks (urban run-off) Favour morphogenic flows Shorten the length of impounded reaches link flood reduction with ecological restoration manage aquatic vegetation
Longitudinal connectivity/continuity improvement	Remove barrier (e.g weir, dam) Install fish pass/bypass for upstream migration Facilitate downstream migration Modify culverts, syphons, piped streams (e.g. daylighting) Manage sluice and weir operation for fish migration Fish-friendly turbines and pumping stations
River bed depth and width variation improvement	Remeander water courses Widen water courses Shallow (i.e. opposite to deepen) water courses Allow/increase lateral channel migration or river mobility Narrow water courses Create low flow channels in over-sized channels
In-channel structure and substrate improvement	Initiate natural channel dynamics to promote natural regeneration Remove sediments (e.g. eutrophic, polluted, fine) Modify aquatic vegetation ('weed') maintenance Introduce large wood Add sediments (gravel, sand) Remove bank fixation Recreate gravel bar and riffles Remove or modify in-channel hydraulic structures Reduce impact of dredging
Riparian zones improvement	Adjust land use (e.g. buffer strips) to develop riparian vegetation Revegetate riparian zones Remove non-native substratum Adjust land use (e.g. buffer strips) to reduce nutrient, sediment input or shore erosion Develop riparian forest
Floodplains/ off-channel/ lateral connectivity habitats improvement	Lower river banks or floodplains to enlarge inundation Set back embankments, levees or dykes Reconnect backwaters (oxbows) and wetlands Remove hard engineering structures that impede laterel connectivity Restore wetlands Retain floodwater (e.g. through local sluice management) Improve backwaters (e.g. morphology, vegetation) Construct semi-natural/artificial wetlands

Table 5.2 shows a detailed composition of each restoration measure type.

5.6 Ecosystem services

The objective of CBA for river restoration is to support decision making in a manner which ensures that those decisions both improve wellbeing and ensure sustainable use of natural resources. To achieve this we need to understand the relationships between those resources and the wellbeing they generate. A framework for shaping and clarifying that understanding is provided by the so-called ‘ecosystem services’ concept.

The Millennium Ecosystem Assessment (MA) defines ecosystem services as “the benefits people obtain from ecosystems” (MA, 2005; p.53). Fisher and Turner (2008) expand on this definition to propose that “ecosystem services are the aspects of ecosystems utilized (actively or passively) to produce human well-being” (p.2051). Both definitions clarify the anthropocentric focus of the ecosystem service concept. While a wider understanding of environmental processes may be a necessary part of any environmental accounting or valuation undertaking, it is the role of the natural world in delivering human wellbeing which is central to the ecosystem service concept. It is this human focus that necessitates the integration of economic analysis within such assessments so that we can quantify and value ecosystem services ensuring that their importance and worth can be incorporated within decision making.

The term ‘ecosystem services’ refers to “those contributions of the natural world which generate goods which people value” (Bateman et al., 2014). We can subdivide these services into two fundamental types: Supporting services are those fundamental ecological functions (e.g. weathering, soil formation, nutrient cycling, etc.), which support all subsequent services. These lie at the base of a potentially extensive and complex chain of further services. ‘Final ecosystem services’ are simply the last item in the chain of natural processes which provides inputs to the production of goods and services used by humans. They are the aspects of the natural environment which most directly affect human wellbeing during an assessment period. While we value the final ecosystem serves rather than the underlying ecological processes, constraints have to be imposed to ensure that ecosystem assets are not run down to unsustainable levels by imposing ecological threshold values and safe minimum standards.

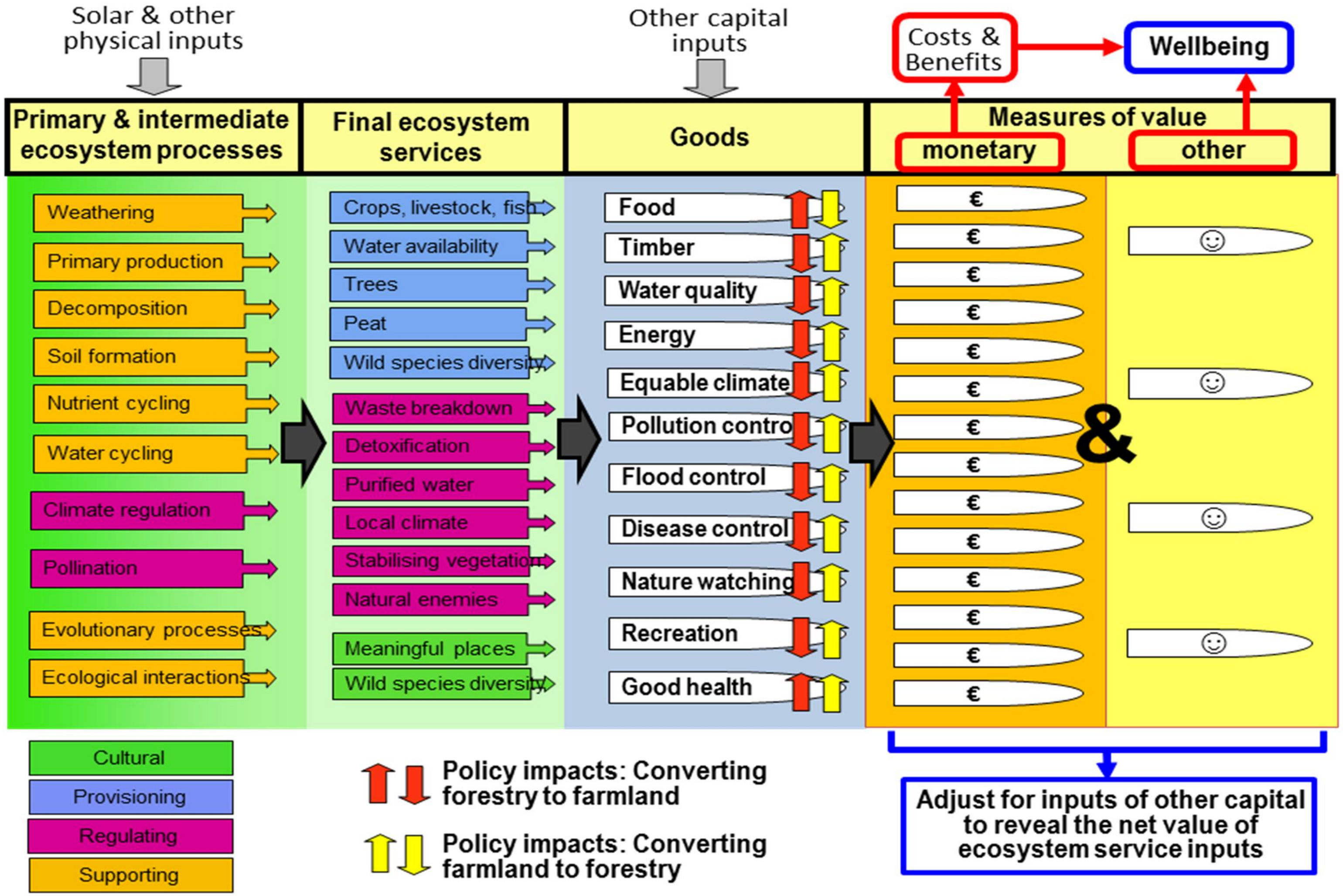
The term ‘value’ is simply the change in human wellbeing generated by a good or service. In economics, the concept of value has a special meaning. In economics, values originate and are measured – where possible – through actual buying and selling behaviour of economic agents on markets. So, values are measured through financial commitment (putting your money where your mouth is) and actual or hypothetical choices (e.g. spending spare time in one way and not the other), since economics is not only about money, but how scarce resources are allocated, including money, time or natural resources.

It is important to note that the same physical good can generate different values to different groups of people, depending among others on its context (e.g. location) and timing of delivery. Some goods generate instrumental ‘use value’ (e.g. timber), while others deliver ‘nonuse value’ (e.g. the knowledge that biodiversity is being conserved even if the person expressing the value does not observe or actively use the species concerned in any way) (Pearce and Turner, 1990).

The value of some goods is reflected in market prices (e.g. timber being sold on the market), although in some of these cases adjustments have to be made where prices are distorted and do not align with prices that would come out of well-functioning economic markets, e.g. where subsidies, interventions, unfair competition, etc. distort prices away from a level which reflects the 'true' underlying value of the goods in question. However, a substantial number of the goods provided in major part by the natural environment lack overt market prices from which economic market values can be inferred. This has led to the development of several methods for estimating values for non-market goods. These approaches are detailed in a wide ranging literature, which has spawned a number of methodological guides (e.g. Bateman et al., 2002; Champ et al., 2003; Freeman, 2003; Kanninen, 2006). These methods facilitate the economic valuation of many non-market goods, which are effectively ignored within simple financial decision-making such as that which directs the majority of private sector production. However, some commentators, while acknowledging the very considerable improvements in broader based cost-benefit economic analyses facilitated by such valuations, also note that there are limits to such valuation. For example, Spangenberg and Settele (2010) and Abson and Termansen (2011) question the use of monetary valuations of the non-use value of biodiversity which rely upon stated preference surveys of public willingness to pay (WTP) for the continued existence of species. They argue that the majority of such studies ask individuals about issues which they have little understanding of (leading to a situation in which the framing of questions can influence responses) or using valuation questions which lack incentive compatibility (i.e. respondents have little incentive to tell the truth about their values). The reliability and robustness of values derived in such ways is debateable. In such cases, alternative approaches based on opportunity costs and objective non-monetary measures of the response of biodiversity to impending changes (such as estimates of the change in affected population numbers) can be used to provide a basis for the search of cost-effective approaches for evaluating biodiversity conservation trade-offs.

A schematic representation of the flow from natural processes through ecosystem services to the delivery of goods and their value is illustrated in Figure 5.6. As noted in the lower right hand corner of the figure, the value of any good cannot be fully attributed to ecosystem service inputs if in fact its production relies in part upon inputs of other capital. However, controlling for the latter allows us to reveal the value of the former (see UK National Ecosystem Assessment, 2011, for examples).

Figure 5.6 also illustrates the analysis of policy alternatives within the ecosystem service valuation approach. Two policies are considered; one positing the conversion of farm land into forestry and the other considering the opposite flow. The diversity and direction of impacts generated by these schemes is illustrated through the arrows placed in the 'goods' column (red arrows for conversions from farming to forestry; yellow for the conversions in the opposite direction). The main message of these illustrations is that a change which is often prompted by just a single good (e.g. an increase in food production) can generate multiple indirect impacts. Furthermore, consideration of those impacts shows that, while a minority have values reflected (often imperfectly) in market prices, many do not. Application of non-market valuation techniques is clearly vital if decisions are to capture the full diversity of values generated by these options. Failure to conduct such valuations is liable to result in incomplete assessments and poor decisions.



Adapted from Bateman et al. (2011), Mace et al. (2011) and UK National Ecosystem Assessment (2011).

Figure 5.6: Conceptual framework for the economic assessment of policies incorporating ecosystem service flows

The restoration of the Mayes Brook in Mayesbrook Park in the London Borough of Barking and Dagenham is part of a broader environmental regeneration project. The restoration of an urban river in a currently barren and unattractive park landscape combines flood storage, biodiversity enhancement and adaptation to climate change within a city environment. The CBA contains a complete ecosystem services assessment of the restoration project, following the MA classification of provisioning, regulatory, cultural and supporting services. For each of 28 specific ecosystem services, it was identified whether the service existed and would be improved by the project, how it could be valued, and, if it could be valued, what its annual value was.

We have considered three main types of Ecosystem Services that can be altered when pressures and water management affect fluvial ecosystems (MEA,2005):

Provisioning services: Products obtained from ecosystems

Regulating services: Benefits obtained from the regulation of ecosystem processes

Cultural services: Nonmaterial benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences

Table 5.3 shows a detailed composition of each Ecosystem Services type.

Table 5.3.- List of main Fluvial Ecosystem Services that are affected by HYMO pressures classified according to Provisioning, Regulating and Cultural types.

Provisioning Service	
Biological raw materials	forestry products poplar plantations genetic resources natural medicines reed and willows used for thatching
Mineral raw materials	drinking water irrigation water construction gravel, construction sand clay for construction, bricks and pottery
terrestrial Food	agricultural dairy and fruit trees crops on terraces
aquatic food	commercial fisheries, Fish yield
Fluvial transport	Fluvial transport
Cooling system	Cooling system
renewal energy	hydropower,

Regulating Service	
Climate Regulation	Local Climatic Regulation Carbon sequestration in riparian woodland
Water regulation	Peak flows reduction flood energy dissipation Soil moisture and aquifer recharge
self-purification	Reduction of organic and inorganic pollutant load Riparian nutrient trap
soil formation	flood retention in floodplain (water, sediment, nutrients) flooding sedimentation
Channel maintenance	Reshaping and adjustments after disturbances
Biological recovery	Dispersion and recolonization mechanims by drift
Biological Control	invasive species control pest/disease control

Cultural Service	
Recreation & Ecoturism	trout and salmon* fly fishing, angling rafting, kayaking, yachting, sailing, sunbathing, swimming, hiking, waterfowl hunting, hunting,
Landscape & Aesthetic values	Scenic beauty of the landscape Nature art
Environmental education	wildlife and biodiversity
Scientific knoledge	
Spirtual & Religious values	

In order to facilitate understanding the effects of pressures and measures on Ecosystem services, we will skip the intermediate steps that include hydromorphological processes and biological responses (see D1.2 and D1.3 REFORM deliverables). We will also avoid multi-pressure analysis and focus on one pressure examples: water abstraction and channelization.

Water Abstraction

Water may be taken directly from the flowing waters in the channel (surface water abstraction), or indirectly from wells by pumping water from aquifers that may be closely connected to rivers (groundwater abstraction). Furthermore, water abstraction from rivers can be achieved through inter-basin flow transfer schemes, whereby the donor river system has its flow reduced because of the diversion.

Groundwater over-abstraction can lead to decline in groundwater levels within aquifers and drying up or causing severe flow reduction in rivers. Surface seepage from aquifers supports groundwater-fed ecosystems such as wetlands and springs. Riparian vegetation affected by declining phreatic levels rapidly shows signs of water stress, leading in extreme cases to widespread riparian plant death.

Figure 5.7 shows the ecosystem services that are affected by water abstraction.

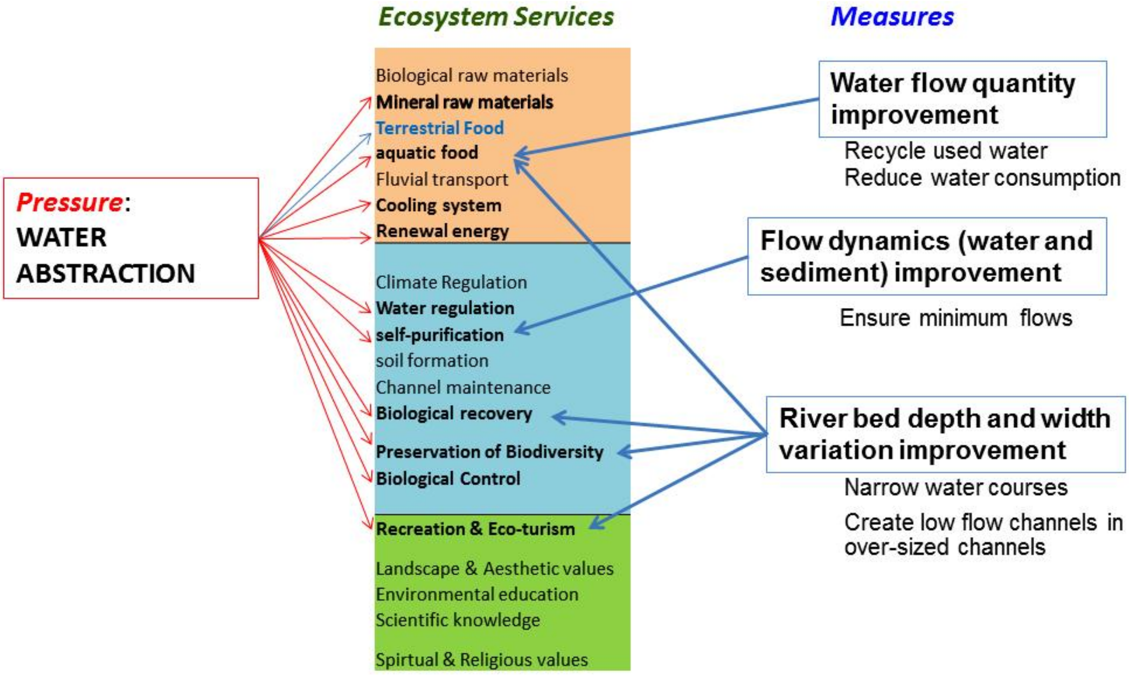


Figure 5.7 Scheme of the interaction among water abstraction and restoration measures and their affections to ecosystem services. Red arrows promote services and blue ones improve them.

Provisioning services are clearly reduced by water abstraction: mineral raw materials (reduction on sediment transport), aquatic food (less habitat for fishes), cooling system (flow reduction) and renewal energy (water flow reduction). However, other provisional service production of terrestrial food is enhanced, as water abstraction is mainly done for irrigation, and also consequent low flow rates allow the cultivation of the river banks.

Among regulating services there is also a clear reduction: water regulation, self-purification, biological recovery, preservation of biodiversity and biological control (all caused by flow reduction). Finally, among cultural services, flow reduction affects mainly recreation and eco-tourism.

In order to mitigate the effects of water abstraction three possible restoration measures may be implemented:

1. Water flow quantity improvement, by recycling used water and reducing water consumption. This measure will directly mitigate the water abstraction pressure, and will also alleviate the reduction of the provisioning of aquatic food service.
2. Flow dynamics improvement, by ensuring environmental flows. This measure will ensure that the worst effects of water abstraction will not take place. Specially, it will enhance the self-depuration capacity of the river.
3. Channel depth and width improvement: creating narrow water courses, and low flow channels to concentrate reduced water flows. This is a structural measure which is no sustainable from the geomorphological point of view. However, while maintained it is likely to improve provisioning services such as aquatic food, cultural services as recreation and regulating ones such as self-purification, biological recovery and preservation of biodiversity.

Looking at Figure 5.7 it can be appreciated that some degraded services are not mitigated by any of the measures proposed. These include mineral raw materials, cooling system, renewal energy, and water regulation. These are the main services on which measures design and research must focus on.

Channelization

'Channelization' refers to river and stream channel engineering undertaken for the purposes of flood control, navigation, drainage improvement, and reduction of channel migration potential. When channelization involves cross section alteration, this includes activities such as channel enlargement through widening or deepening, the reduction of flow resistance through clearing or snagging of riparian, and sometimes aquatic, vegetation and other roughness elements, and the introduction of bank facing and reinforcement materials. These forms of morphological modification typically transform channel cross profiles into uniform, smooth, trapezoidal or rectangular forms. Cross section alteration can also include embankment, levee or dyke construction, which further enlarge the channel capacity, prevent channel-floodplain connectivity, and can induce very high flow velocities within the river channel during floods.

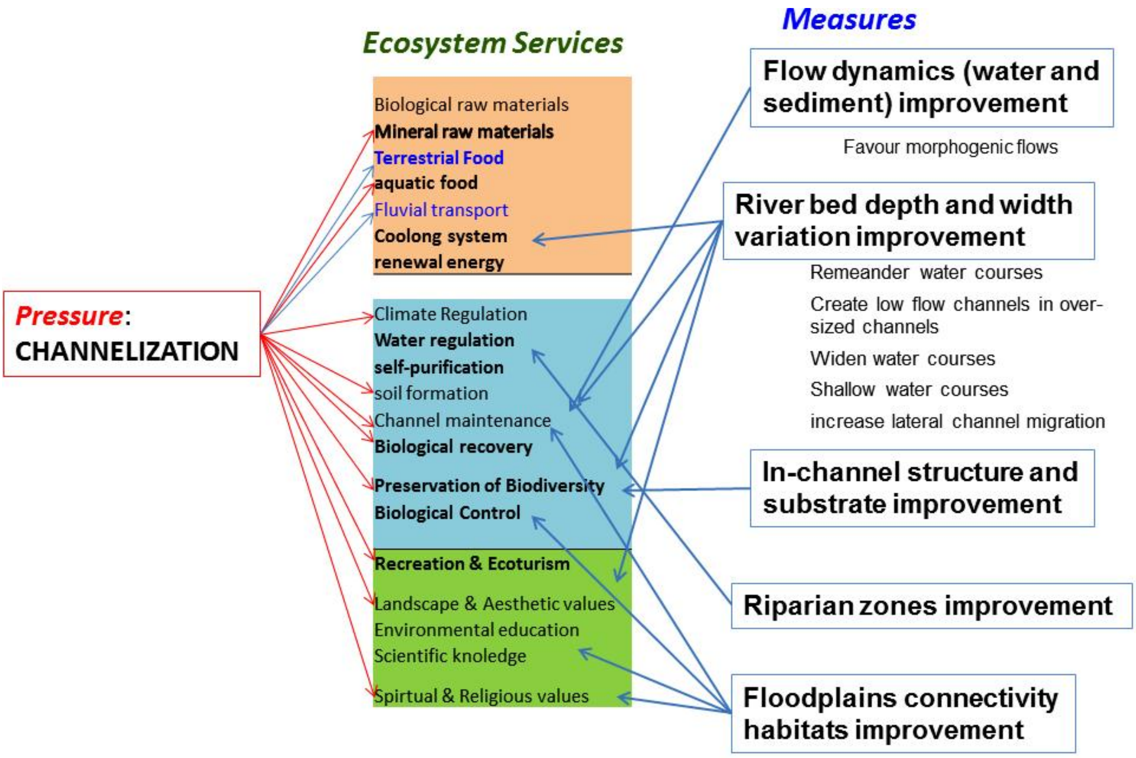


Figure 5.8 Scheme of the interaction among channelization pressure and restoration measures and their affections to ecosystem services. Red arrows promote services and blue ones improve them.

Figure 5.8 shows the ecosystem services that are affected by channelization (cross section alteration).

Provisioning services are reduced by water abstraction: mineral raw materials and aquatic food (less habitat for fishes). However, other provisional services are enhanced: production of terrestrial food, as channelization protects floodplain farming from flooding; and fluvial transport because of increased water depth.

Among regulating services there is a clear reduction: climate regulation, soil formation (as the floodplain is prevented from flooding), channel maintenance (levees and lateral embankments promote channel incision), biological recovery, and preservation of biodiversity (both caused by habitat homogeneity and refuge loss). Finally, among cultural services, channelization affects recreation and eco-tourism, landscape and aesthetic and spiritual values are all degraded.

In order to mitigate the effects of channelization five possible restoration measures may be implemented:

1. Flow dynamics improvement, by ensuring flushing floods. This measure will ensure channel maintenance though geomorphological processes.
2. Channel depth and width improvement: creating lateral gentile slopes, low flow channels and giving space to the river. These structural measures are not sustainable per se. However, while maintained they are likely to improve services such as aquatic food, channel maintenance, a good habitat for biodiversity and landscape and aesthetic values

3. In-channel structure and substrate improvement are also not sustainable measures, but will enhance biological recovery and biodiversity.
4. Riparian Zones improvement will improve water regulation capacity as it promotes infiltration and aquifer recharge during spates.
5. Floodplain connectivity: this is a great measure against channelization because it regulates water during floods, controls biological processes linked to the riparian system, and also enhances possibilities of environmental education and scientific knowledge.

The analysis of HYMO pressures that impact fluvial ecosystems and restoration measures that improve ecosystem services has been done in a qualitative manner and under strong assumptions. The lack of a strong scientific basis linking the physical characteristics of rivers with their natural ecosystem functioning and ecosystem services makes quantitative predictions difficult (Thorp et al., 2010). The decision-making on river restoration implementation needs this prediction capacity, as it is going to be negotiated among economic interests, environmental issues and sociopolitical factors. Specially, if we want to take decisions based on cost benefit analysis, the first step will be to assign monetary values to ecosystem services. If this is so, the conclusion is clear that more quantitative research is needed: a holistic research that include HYMO processes, biological responses and the economic balance of the associated ecosystem services.

5.7 Non-market valuation of benefits

Economists have developed a number of methods for estimating the value of goods and services whose market prices are either imperfect reflections of that value or non-existent. These methods are designed to span the range of valuation challenges raised by the application of economic analyses to the complexity of the natural environment. Guidelines on their application are available in detail in a number of existing reviews. One particular useful guideline for applying these methods to the appraisal of river restoration projects is the UK Environment Agency's recent Water Appraisal Guidance (Shamier et al. 2013) that was discussed in Chapter 2 of this report. The selection of the appropriate valuation method is partly determined by the ecosystem service being valued. Table 5.4 indicates which primary valuation methods can be used to value each ecosystem service.

Table 5.4 distinguishes between five types of methods: market price based, production function, cost-based, revealed preference and stated preference approaches. The value of ecosystem services with a commercial value (for example crops, timber) is usually capitalized in the value of land on which they grow. Market prices, possibly appropriately adjusted, can be used to value such ecosystem services. Some ecosystem services provide a unique input to some industry, for example wild fish to the capture fishing industry. The production function approach estimates the marginal value of the fish as input into the fishery industry, just as a normal company would estimate its willingness to pay for an extra unit of input of some product or service (e.g. one kWh of electricity, or one m^3 of drinking water). Replacement/restoration cost is widely used as a measure of ecosystem service value. Estimations of cost, however, are generally not good proxies for benefits. The underlying assumption for this valuation method, which may not always be valid, is that the benefits are at least as great as the costs involved in replacing or restoring an ecosystem service. The replacement/restoration cost method will tend to over-estimate ecosystem service values if society is not prepared to pay for man-made replacements (i.e. if there is insufficient demand). Alternatively, in the case that society is prepared to pay for the man-made replacement, the cost of replacement provides only a lower bound estimate of the benefit (i.e. we only know that the benefits of restoration exceed the cost).

An example is given in the box below where public WTP for ecological river improvements in Switzerland was estimated based on the recently revised Water Protection Act, which allocates 5 billion CHF to rehabilitate 4,000 km of the most degraded 15,800 km of Swiss rivers within the next 80 years. Although this was a decision by the Swiss parliament, it has been argued that this value estimate can be used as a lower bound for public WTP in Switzerland as it resulted from a democratic process initiated by the Swiss fishing association, taken over by the parliament, and not questioned by the population. This indicates that there is strong public support for this funding. However, as it was not inquired by a survey (see the other example below), we only know that this funding is supported and do not know, whether the WTP of the population would be higher. It is thus considered as a lower bound for the WTP for ecological river improvements for river sections not influenced by hydro-power plants.

In 2005, the Swiss fishing association launched a popular initiative on the protection of water bodies. In the following year, it got the necessary number of signatures for bringing this topic to a public vote. The main elements of this constitutional initiative included the advancement of the revitalization of water bodies, measures against hydropeaking, the reactivation of the bed-load balance, and strict enforcement of the regulations on residual waters. The initiative also proposed to give the right to fishing as well as environmental organizations on petition and appeal, which would have enabled these organizations to enforce environmental measures through legal means. However, the federal parliament developed a counterproposal that included most of the elements of this initiative. As the proposal fulfilled most of the requirements of the initiative and there was no referendum against this proposal, the Swiss fishing association decided to withdraw its initiative in 2010 before it came to a public vote. The counterproposal subsequently came into effect as the revised water protection act in 2011. The difference between the proposed initiative and the revised water protection act is that the act allows for more flexibility in the enforcement of the regulations on residual waters and it does not include the right on petition and appeal of fishing and environmental organizations.

The revised water protection act aims at rehabilitating 4’000 km of rivers of the 15’000 km that are most strongly degraded. These efforts aim to rehabilitate natural water structures, enable fish migration, and ensure the natural diversity or the rivers. The costs of these revitalization efforts were estimated based on the average costs of revitalizations per kilometer, including land acquisitions. The costs of removing artificial impediments to fish migration were also added to this cost estimate. The total costs of these measures sum up to roughly 5 billion CHF. Due to the financial and temporal magnitudes of this strategy, it was decided to conduct the revitalization during a time-period of 80 years. This results in annual costs of about 60 million CHF until 2090. The Swiss federal government has committed to cover 65% of the annual costs (i.e. 40 million CHF/year), while the rest has to be covered by individual cantons in which river restoration measures take place. The responsibility for implementing the restoration measures (as well as for identifying stretches of rivers to be treated with restoration measures) lies at the cantonal level but implementation must be according to the federal guidelines to get the federal funding.

In addition to these river rehabilitation efforts, the revised water protection act also intends to reduce the negative influence of hydropower plants regarding hydro-peaking and, water diversion, gravel transport, and fish migration, rehabilitate lake shores, and secure the spatial requirements of rivers and lakes. These measures have different funding mechanisms as has flood protection which is often combined with rehabilitation.

The revealed and stated preference methods infer the value of an ecosystem service from observed economic behavior of people or from surveys. Examples of revealed preference methods are for example the Hedonic Price method and the Travel Cost method, in which house prices or the travel behaviour of people ‘reveal’ the preferences of house buyers or recreationists for associated ecosystem services. Currently, among the stated preference methods, the Choice Modelling method is the most frequently applied method. With this method, sampled respondents are asked to make trade-offs between carefully described ecosystem services and other goods and services to elicit their WTP for these ecosystem services.

A study to elicit the preferences for ecological restoration of heavily modified river stretches of the Danube river, the second largest river in Europe, carried out Choice Modelling experiments in three countries; Austria, Hungary and Romania. About 1500 respondents were offered choices of two exclusive categories of benefits: the impact of river restoration on floodwater storage and a corresponding reduction in flood risk, and the river's nutrient retention capacity and hence water quality. A monetary cost price was included as a third attribute to enable monetization of the benefits of different river restoration projects. The alternatives describe different end states created through river restoration measures. Variations in end states are caused by different degrees of river restoration and corresponding scale effects. In this way, respondents were not asked to value the river restoration measures per se, but rather their outcomes in order to avoid correlation due to causality. To increase the realism of the presented alternatives, respondents were shown existing river restoration plans on a map. The three main conclusions of the study were: 1) the WTP for the river restoration plans differed greatly among the three countries and the potential for transferability between the countries is limited, and 2) aggregation errors are large when average values are aggregated without controlling for preference heterogeneity (due to distance-decay and income effects) and socio-economic conditions that are unevenly distributed in space. (Brouwer et al., 2009).

Primary valuation involves estimating the value of ecosystem services through the collection of data that is specific to the ecosystem(s), service(s) and beneficiaries that are under consideration. An example is given in the box below, based on a survey carried out in Finland.

In Finland approximately 55% of the peatlands have been drained for forestry to increase forest growth (Turunen 2008) and dense ditch networks currently characterize the forested landscape causing increased organic and inorganic sedimentation and suspended solid, nutrient and metal loading to streams (Liljaniemi et al. 2003). Sedimentation homogenizes the natural stream bed habitat by burying cobbles, boulders, dead wood and bryophytes, and can have detrimental effects on the native biota such as macroinverbrates and fish (Suurkuukka et al. 2014). Few river restoration projects have aimed to mitigate forestry impacts, although the habitat degradation may have a long-lasting (up to 40 years) impact (Zhang et al. 2009). A notable exception are forested streams in the River Iijoki catchment in North-Eastern Finland, where The Finnish Forest and Park Service has restored approximately 45 stream kilometers in 31 streams in the past sixteen years (Luhta et al. 2014). The main aim of these restorations has been to mitigate the impacts of the forestry on the streams.

In order to assess the societal benefits of river restorations, a Contingent Valuation (CV) study was carried out in Reform to elicit and measure attitudes of residents and forest owners towards stream restoration in the River Iijoki catchment in the Koillismaa region. A questionnaire survey was conducted to assess how residents and forest owners in Koillismaa value their forest streams and stream related ecosystem services. Besides a more qualitative assessment of opinions and attitudes, the benefits of stream restoration were evaluated by estimating individual households' maximum willingness to pay for the Koillismaa Forest Stream Restoration and Maintenance Program. The pictures below were used to demonstrate the situation without (left) and with (right) the restoration program in place. These CV results provide valuable input for a CBA when determining whether the restoration program is socially desirable and thus worth implementing. To this end, a questionnaire was mailed by the end of 2013 to 1,764 randomly selected households living in the municipalities located in the Iijoki river catchment. Almost 40% completed and returned the questionnaire.

The results show that respondents appreciated various ecosystem services provided by the Koillismaa forests and streams. The most appreciated services were clean and fresh air (75%), clean waters (74%) and the natural products provided by the forests such as berries and mushrooms (58%). Also the landscape and the sounds of the forest were highly appreciated. Most appreciated activities include berry and mushroom picking (62%), fishing (52%) and other outdoor activities (49%). A considerable share of the respondents also consider the protection of threatened species such as the freshwater pearl mussel (34%) and generally the protection of biological diversity (26%) important. See also the Table below.

Almost 70% of the respondents indicated to be willing to pay for the improvements of stream quality. In addition, just over 60% of the respondents were willing to carry out voluntary restoration works in the forest streams in Koillismaa. The mean WTP in the study area was between 21 and 35 Euros per household per year. If we aggregate this WTP estimate across all households living in the municipalities of the study area (22,000) over the period 2014-2018 under the assumption that non-respondents have the same preferences and WTP as respondents, aggregate WTP reflecting the flow of ecosystem services provided by the forest stream restoration program would amount to 92,000-156,000 Euros annually or 0.5-0. 8 million euros in total over the period 2014-2018.

This benefit estimate can be compared to the costs of stream restoration. Stream restoration projects have largely been funded by the Ministry of Employment and Economy in Koillismaa to decrease unemployment. The target presented in the questionnaire was to restore 200 streams in the Koillismaa area during the next five years. It has been estimated that the restoration need would be approximately 240 kilometers in practice (Luhta 2014). Based on an average restoration price of 15 euro per meter stream, the total restoration costs of 240 kilometers would be 3.6 million euro. In this case study the cost of restoration exceeds the benefits gained by the residents from the restoration, mainly due to the fact that the area is sparsely populated. The benefit-cost ratio for stream restoration is therefore lower than one and stream restoration does not seem to generate net benefits for the population of beneficiaries in the area.

Value transfer, by contrast, involves estimating the value of ecosystem services through the use of value data and information from other similar ecosystems and populations of beneficiaries. It involves transferring the results of primary valuations for other ecosystems (“study sites”) to ecosystems that are of current policy interest (“policy sites”). Reliable value transfer is dependent on the availability of reliable primary valuation results. It is not a perfect substitute for primary valuation. As the number and breadth of reliable primary valuations increases, the scope for reliable value transfer also increases. Ecosystem service values estimated using value transfer may be characterized by high uncertainty. For this reason it is preferable to conduct primary valuations of ecosystem services, if resources (data, time, expertise, knowledge) permit.

In cases where it is not possible to produce sufficiently robust value estimates, either through primary valuation or value transfer, we have to accept that for some ecosystem services and contexts it is not possible to estimate monetary values for the associated welfare.

The Aln and Coquet Fish Pass Improvement project in Northumbria consists of a suit of projects aiming at fish pass improvements and improving the quality of 11 water bodies from moderate to good in the WFD classification. Because of the project, salmon will have access to an additional 192 km of river for spawning habitat. It is expected that the project will deliver both use and nonuse benefits. The use benefits are for recreational anglers who see their chances of catching salmon increase and the nonuse benefits are for the local population. For the valuation of these benefits, ‘value transfer’ was used. The use benefits were based on a study by Davis and O’Neill (1992) on the WTP for angling in Northern Ireland. The original estimate of WTP per angler per trip was inflation adjusted. Total use benefits were calculated by multiplying this inflation-adjusted WTP number by the estimated number of current angling trips. Nonuse benefits were based on a study by Spurgeon et al. (2001) who estimated an annual WTP per household (of non-anglers) for maintaining or improving fish populations in their most familiar water body. Applying the Distance Decay Method of the Environment Agency’s Benefits Assessment Guidance (2003), a population within 60 miles of the center point of the project was selected. The adult population was divided by the average number of household members to derive the number of households. The inflation-adjusted annual WTP per household was multiplied by the number of households to arrive at the total annual nonuse benefits.

Table 5.4 Ecosystem services and applicable valuation methods (Source: Brouwer et al. 2013)

Services	Valuation methods	Comments
Provisioning		
Crops/timber		Most ecosystem services of agro-ecosystems will be capitalized in land prices. They should be adjusted for specific capital investments, such as for irrigation and drainage.
Livestock		
Wild foods		The market price of a close-substitute food or fuel might be a fair proxy.
Wood fuel		
Capture fisheries		The production function method is preferred, see Barbier (2007). Otherwise (adjusted) market prices can be used as a rough proxy
Aquaculture		
Genetic		Appropriate market prices are for example license fees for prospecting. An alternative valuation method is based on the costs of alternatives approaches to recover genetic information.
Fresh water		Market prices (if available), shadow prices (through production function method). If there is strong evidence for demand fresh water, cost of alternative supply
Regulating		
Pollination		If there is strong evidence of demand for pollination services, expenditure on alternative pollination technologies (replacement cost) might be used. Avoided cost is an alternative.
Climate regulation		The preferred cost-based method is 'damage cost avoided'
Pest regulation		If there is strong evidence of demand for pest regulation, expenditure on manufactured pest regulation products (replacement cost) might be used
Erosion regulation		The preferred cost-based method is 'damage cost avoided'

Services	Valuation methods			Comments
Water regulation				Avoided expected damage costs of floods and droughts; revealed or stated preference methods might be used to estimate the willingness to pay to avoid these expected damages
Water purification				If there is strong evidence of demand for clean water, replacement cost might be used (see e.g. Chichilnisky and Heal, 1989)
Hazard regulation				Avoided expected damage cost; revealed or stated preference methods might be used to estimate the willingness to pay to avoid these expected damages
Cultural				
Recreation				Methods include travel cost methods, contingent valuation, choice experiments
Aesthetic				Methods include hedonic price methods, contingent valuation, choice experiments

	Market price based methods ((adjusted) market prices, net factor income,)
	Production function methods
	Cost-based methods
	Revealed preference methods (travel cost method, hedonic price methods)
	Stated preference methods (contingent valuation, choice experiments)

5.8 Indirect effects

Restoration projects often have effects beyond the direct effects of the project on the objectives it sets out to achieve and the associated beneficiaries. Indirect economic effects are the spin-off effects of the market transactions of the owner, operator or user of the project services. These may manifest themselves on other markets and market prices or remain unpriced because they are public in nature. For example, in urban river restoration projects, the surrounding neighbourhood may become more attractive, affecting house prices and rentals, thereby indirectly affecting people who do not make use of the primary project services (see the example in the Box below). These indirect effects are important in evaluating investments in public goods or infrastructure, because they determine the final distribution of the costs and benefits of a project. This does not mean, however, that the indirect effects can always be added to the direct effects to determine the total utility of a project. The government guideline on cost-benefit analysis of infrastructure projects in the Netherlands argues: *"A stone thrown into a calm pond causes ever wider ripples, but the eventual rise of the water level is still equal to the volume of the stone"* (translated from Elhorst et al., 2004).

In the CBA of the restoration of the Mayes Brook in Mayesbrook Park in the London Borough of Barking and Dagenham it is argued that the prices of adjacent property will capture, or at least act as a market surrogate, for a suit of diverse benefits that are associated with an improvement in social relations in the neighbourhood of the Park. This effect on house prices can be construed as an 'indirect effect' (and not likely to be 'additional'), but it is in this case appropriate to include this indirect effect in the CBA, because it is, as explained, used as a proxy for direct benefits and there is no fear of double-counting.

Another example is the attractiveness of a restored river site to recreational users who are drawn away from other sites along the same river. The impact of the river restoration is in that case not overall incremental, the observed indirect effects represent so-called redistribution effects, they do not generate more welfare, they simply redistribute the same level of enjoyment from one site to another. Only if more visitors would be drawn to the river, one could speak of an incremental effect. But the question then becomes where these incremental numbers of visitors come from. If they would have recreated elsewhere and were involved in similar recreational activities as along the restored river, at a larger scale one could still be dealing with a redistribution effect. Hence, the identification of the spatial scale of the cost-benefit analysis is of great importance here to identify whether certain indirect effects are directly attributable as incremental costs or benefits of a restoration project. These substitution effects in the context of the Water Framework Directive have been investigated in detail in another European research project called AquaMoney (see Schaafsma et al., 2013).

Whether and to what extent indirect benefits of river restoration projects occur and whether they are additional to the direct benefits, is a difficult question that can often not be answered quickly. The identification and assessment of indirect effects requires a fair amount of knowledge and information about possible redistribution or indirectly attributable incremental costs and benefits and this is often difficult in practice. Without convincing evidence of the opposite, the default assumption in CBA's of river restoration projects should be that indirect effects do not affect the magnitudes of the costs and benefits, and thus do not affect the net present value or cost-benefit ratio of the project.

5.9 Disproportionate costs

In the evaluation of WFD-related projects, the term 'disproportionate costs' is sometimes used. 'Disproportionate costs' is a rather vague concept in the WFD that allows derogation, i.e. to lower environmental objectives or delay them in time based on (1) technical feasibility of achieving the objectives and/or (2) disproportionate costs (paragraphs 3–7). Here the focus is on the latter criterion. The concept of disproportionate costs is not a standard economic concept. The assessment of disproportionate costs is subjective (Water Economics (WATECO) Guidance Document, European Communities 2002), depends on the political economy of a country or river basin region, and proves to be surrounded by a great deal of uncertainty as to its exact definition and measurement scale in the practical implementation of the WFD (Brouwer, 2008).

From an economic perspective, CBA is the obvious tool to assess disproportionate costs, using either the net present value or benefit-cost ratio as a key decision criterion, comparing all positive and negative welfare effects of the WFD measures to achieve good chemical and ecological status. Assuming that all positive and negative welfare effects are properly accounted for in the CBA, a negative net present value (NPV) or a benefit-cost ratio less than one, indicating a welfare loss, would result in principle in a rejection of the proposed programme of measures to reach the WFD objectives. However, given the public good nature of the WFD objectives and in the interest of future generations, the government may be willing to accept a benefit-cost ratio of less than one. The obvious question would then be by what fraction the costs can exceed the benefits before the costs are considered disproportionate. Uncertainty plays an important role here too, related to the assessment of the costs and the environmental benefits of WFD implementation. Investment costs in water quality improvement are often surrounded by less uncertainty than the associated environmental benefits (Brouwer and DeBlois 2008). An important question then is how much uncertainty policy makers are willing to accept to financially commit to substantial investments whose returns are uncertain (Brouwer 2008).

Since the WFD does not define disproportionate costs (neither do the WATECO guidelines), the definition is left to the EU Member States. Görlach and Pielen (2007) present different perspectives that exist in Europe. In general, there seems to be consensus that the term disproportionate implies that costs are disproportionate in relation to either the benefits of WFD implementation or the available financial resources. In the first case disproportionate implies that the implementation of the WFD – even in the least cost way – is not economically efficient. If costs exceed benefits, the welfare impact of WFD implementation is negative and it would be better to spend the limited available resources on less ambitious environmental objectives or to spend them in an alternative way altogether, for example, on health care, education or employment. However, a technical issue here is that a CBA requires that all costs and benefits of WFD implementation can be assessed and estimated in monetary units, which may not always be the case.

Comparing the costs of WFD measures with the available financial resources, a different type of indicator for disproportionate is found. Even an economically efficient policy may not be affordable if the necessary financial resources are not available. In most Member States lack of financial resources is primarily a matter of public budget allocation and hence a political decision. Affordability is expected to be a completely different matter at the level of individual households and economic sectors. In the latter case the competitive position and hence the economic survival of a sector may be at stake depending on the sector's financial solvability and viability. In the former case, lower-income households may see their purchasing power reduced to socially and politically unacceptable (poverty) levels. Benchmarks and thresholds will have to be defined at European level on the basis of socially and politically defined 'acceptability' criteria, taking into account differences between MS. For example, the total water bill cannot exceed more than 5% of total disposable household income. If measures are not affordable for a certain sector or group of households, financial transfers can be used to share the burden. This may also be welfare enhancing, as total welfare improves if costs are re-distributed from the relatively poor to the relatively well-off given decreasing marginal utility of income.

In France, water managers seem to attach most importance to affordability type of arguments that underline the importance of social factors in defining disproportionate costs. One of the French Water Agencies proposes, for example, a threshold of a maximum of 20% increase in absolute costs to screen potentially disproportionate measures, whereas another French Water Agency holds on to an absolute limit of maximum 2% of total household expenditures (Görlach and Pielen 2007). In the UK, the balance between costs and benefits is considered more important, and attention focuses mostly on the comparison of costs and benefits. Financial arguments are also important, but only after comparison of total costs and benefits. The Netherlands is somewhere in between these two approaches, and both welfare economic and affordability arguments play a role (Brouwer, 2008). Most importantly, the Netherlands underline the political character of the disproportionate cost discussion, suggesting that it is up to policy makers to define what thresholds should be used. For this, several indicators are suggested at different relevant levels, such as costs as a percentage of GDP or the percentage increase of the water bill compared to disposable household income.

Clearly, for the European WFD to be effective it is important that MS use comparable criteria in defining disproportionate costs and that the methods and indicators used are transparent and clear. Also for the more limited use of disproportionate costs in the evaluation of river restoration projects, there is an obvious need for clear and transparent criteria.

6 Conclusions

This report identified and addressed some of the key issues related to the assessment of the costs and benefits of river restoration projects, directly related to the different steps taken in conducting a Cost-Benefit Analysis (CBA). The issues were illustrated as much as possible with examples from practical case studies.

The report started with identifying existing manuals and guidelines for the economic analysis of river restoration projects. It was concluded that while such manual or guidelines do not yet exist, there are a number of important guidelines on the economics of water management in general that also offer valuable advice to experts involved in the assessment of economic costs and benefits of river restoration projects. The work presented in this report provides an important supplement to and/or extension of these existing guidelines, in particular the water appraisal guidance (Shamier, 2013).

Costs include financial and (non-financial) economic costs, including external costs. A relevant classification of cost categories is the standard WFD-related cost typology which was developed for the cost-effectiveness analysis of the Programs of Measures (PoMs). It distinguishes between non-recurring costs, recurring costs, non-recurring and recurring costs for regulators, cost savings, transfers, non-water environmental costs or benefits resulting from implementing a measure, and wider economic effects. There are also other classifications, and it is important to use the classification that best fits the purpose of the cost assessment (e.g., financial or social economic). Cost estimates can be used in decision-support tools such as CBA or Cost-Effectiveness Analysis (CEA). In the practical examples of the economic assessment of river restoration projects in the USA, CEA played an important role in the process of selecting restoration measures. In Europe, experiences with the prioritization of restoration measures in the context of the WFD based on CEA are still very limited. In the US example, the CEA aided in prioritizing restoration measures and plans, developing cost effective combinations of measures and eliminating cost ineffective plans. CEA is a useful tool for both small-scale and large-scale restoration projects and applicable under consideration of a very broad range of restoration measures.

In order to be able to apply CBA, also the benefits of river restoration have to be assessed. River restoration provides a wide array of hydrological, ecological and socio-economic benefits. Many of these benefits are so-called public goods and services provided by restored or natural river systems, and can only be estimated in monetary terms using non-market valuation techniques. Economists have developed a number of methods for estimating the value of goods and services whose market prices are either imperfect reflections of that value or non-existent. Primary valuation involves estimating the value of ecosystem services through the collection of data that is specific to the ecosystem(s), service(s) and beneficiaries that are under consideration. Primary valuation methods include market price based methods, production function methods, cost-based methods, revealed preference and stated preference methods. These valuation methods each have their advantages and disadvantages, and some are more appropriate to assess the value of certain ecosystem services than others. One thing they have in common, is that their application is often time-consuming and relatively expensive. Therefore, in many cases, value transfer methods are applied instead.

Value transfer involves estimating the value of ecosystem services through the use of value data and information from other similar ecosystems and populations of beneficiaries. It involves transferring the results of primary valuations for other ecosystems ("study sites") to ecosystems that are of current policy interest ("policy sites"). Reliable value transfer is dependent on the availability of reliable primary valuation results. This report carried out a meta-analysis of 39 primary river restoration valuation studies from across the world. After standardizing the monetary variables of the studies to a common base year (2013) and a common currency (the Euro), the willingness-to-pay for river restoration was regressed against a number of explanatory variables including river and location characteristics, population characteristics, ecosystems services and study characteristics. The average willingness-to-pay for river restoration across the studies was found to be approximately €70 per household per year. It was concluded, however, that for benefit transfer purposes it would be better (more accurate) to transfer a benefit *function*, including local characteristics on the extent of restoration, population density, per capita income, and individual ecosystem services to be restored. This report provides the parameters (coefficients) of such a benefit transfer function.

There are still a number of methodological problems in carrying out a CBA of a river restoration project. The key problem for CBA as well as for CEA and other decision-support tools is arguably the *ex ante* assessment of the causal chain from restoration measures via ecological impacts to the delivery of final ecosystem services. Other problems include the definition of the baseline ('what would happen with ecological quality and ecosystem services over time *without* the restoration measures'), the assessment of indirect effects and disproportionate costs. Other potential problems include the classification of cost categories (particularly financial versus economic) and their assessment, and the non-market valuation of benefits. This report has aimed to provide support for addressing these problems.

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